UNDP/GEF PROJECT ENTITLED "REDUCING ENVIRONMENTAL STRESS IN THE YELLOW SEA LARGE MARINE ECOSYSTEM"

UNDP/GEF/YS/WRI.1/3
Date: 23 October 2007
English only

Proposal and Report Writing Workshop for Environmental Practitioners:

Ansan, ROK, 22-23 October 2007

Keys to Effective Writing

Report of the
Proposal and Report Writing Workshop for Environmental Practitioners: Keys to Effective Writing

Summary of the Workshop

The UNDP/GEF Project on "Reducing Environmental Stress in the Yellow Sea Large Marine Ecosystem (YSLME)" organised the "Proposal and Report Writing Workshop for Environmental Practitioners: Keys to Effective Writing" in Ansan, Republic of Korea (ROK), 22-23 October 2007, to strengthen the capacity of government agencies and research organisations to prepare high-quality proposals and reports for securing the integrity and sustainability of environmentally-related research activities that the organisations implement. Eighteen professionals who deal with marine and coastal management participated in the Workshop: ten from China and eight from ROK. A professor from the Indiana University at Bloomington was invited to serve as the workshop lecturer. The lecturer has expertise in freshwater and estuarine wetlands ecology and a proven track record of publishing dozens of peer-reviewed journal papers. A list of the participants and lecturers is attached as Annex I to this report.

The workshop, consisting of lectures and hands-on exercises, covered how to write effective proposals, research papers, and abstracts. The lectures described tips for writing proposals, including proposal organisation, experimental design and statistics, and presentation of data (figures and tables). The hands-on exercises illustrated how to present data as figures and tables and how to organise and list references (in-text citations and bibliographic list). Tips for writing a research paper, including the structure of the paper, composing the abstract and use of SI units, were also covered. A writing exercise to compose abstracts for presentations, proposals and research papers was conducted as part of the hands-on exercises.

The workshop was conducted in English. All the teaching materials used during the workshop are attached as Annex II in this report.

1. Objective of the Workshop

1.1 The objective of this workshop was to provide officials of both governmental and non-governmental research organisations with a set of instructions and interactive writing exercises to prepare high-quality proposals and reports.

1.2 Throughout the workshop, it was expected that the participants would develop a good understanding of not only the basic structure and elements of a proposal and a report, but also the keys to writing them effectively.

1.3 After completion of the training, the participants were expected to contribute to securing the integrity and sustainability of environmentally-related research activities that are implemented under the Project as well as those that other relevant organisations implement.

2. Contents of the Workshop

2.1 The workshop consisted of two parts: Part I for proposal writing and Part II for report writing. Specific lecture topics included the following:

Part I - Writing Effective Proposals

- Structure of a typical research proposal
- Function of the subsections and what a typical research proposal should contain
- Statistical analysis and experimental design
- References (in-text citations and bibliographic list)
- Effective presentation of data, including the use of tables and figures
- Writing exercise: Organizing and presenting data in tables and figures. Preparation/presentation of references

Part II - Writing an Effective Report

- Structure of a typical research paper
- Organisation of the abstract
- Acknowledgement of sources (intellectual property and plagiarism)
- Use of SI (System International d'Units) International System of Units (the metric system)
- Writing Exercise: Composing abstracts for presentations, proposals, and research papers

2.2 To get the most out of the workshop, the participants were requested to bring data they had collected and analysed statistically, and a partially completed proposal, research paper or abstract to work and improve upon. For the writing exercise, the participants were divided into small groups with two to three members.

2.3 Mr. Christopher Craft, Professor, Indiana University at Bloomington gave lectures and instructed group exercises. Professor Craft is President-elect of the *Society of Wetland Scientists* (2007-2008), a 3,500 member international organization, and past-president of Division S-10, Wetland Soils, of the *Soil Science Society of America*. At the university, Professor Craft teaches courses in Wetlands Ecology, Restoration Ecology, Applied Ecology and Environmental Science, advising undergraduate, Master's and PhD student research. Mr. Craft and his students have published more than 60 peer-reviewed research papers and given more than 100 presentations at national and international meetings. Professor Craft holds a PhD in Soil Science from North Carolina State University.

2.4 In addition to the above lecture topics, Mr. Craft also gave lectures on Environmental Monitoring, explaining its importance and use for environmental policy-making. Additionally, Professor Craft discussed ecological issues in China, focusing on Yangtze River flooding, wetland restoration, and Three Gorges Dam.

3. Key Points of the Lectures

3.1 This section summarises the key points of the lectures provided by Mr. Craft.

Abstract writing

3.2 A good abstract should address the following four questions in order:

1. What did you do?
2. Why did you do it?
3. What did you find?
4. Why is it important?

3.3 It is important to include some data in the abstract of reports. The abstract of proposals would not have data, or it may have some preliminary data.

3.4 An abstract should be composed to attract people to read further. Note that people often look at the abstract first, and may not read the whole paper.

Proposal/report writing

3.5 There are some commonalities in writing proposals and reports. For example, proposals and reports have a similar structure with common components such as abstract, introduction, and methods.

3.6 However, there are differences, too, between these two kinds of documents. For instance, proposals describe future research activities, while reports describe past/completed research activities.

3.7 Table 1 below summarises major components of proposals and reports with tips for effective writing, articulating the commonalities and differences between the two documents.

Table 1. Structure of Proposals and Reports

Proposal **(for future activities)**	**Report** **(for past/completed activities)**
Abstract/project summary • Use all of allowed length. • Address the four questions: (1) What did you do? (2) Why did you do it? (3) What did you find? (4) Why is it important?	**Abstract** • Be concise. An abstract for journal article is shorter than that for proposal. • Address the four questions.
Introduction • Mention peer-reviewed literature to find out what has already been done in the same field.	**Introduction** • Mention peer-reviewed literature to find out what has already been done in the same field.
Hypotheses • Isolate hypotheses from text to show them clearly. • Use a conceptual diagram or “cartoon” for easy understanding.	**Objectives (hypotheses)** • Put objectives at the end of “Introduction.” • State them in two to three sentences.
Methods • Include site map, statistics, tables, and/or models.	**Methods** • Include site map, statistics, tables, and/or models.

Relevance/significance • Refer to Call for Proposals and highlight why your proposal is relevant to their request.	**Results** • Show data only; do not discuss about data in this section. • Use figures to show trends and/or differences. • Use tables to present a lot of data.
Synergies • Synergies with other research (How will you link to other similar research? Is there international or national collaboration?)	**Discussion** • Relate to "Introduction" (How do your findings relate to previous work by you and others?). • Discuss whether your findings are the same or different from previous studies; if different, explain why.
References	**Conclusions** • Describe what you found and why it is important in one concise paragraph. • Have some similarity to Abstract; address the questions # 3 and #4 (i.e. What did you find? Why is it important?). • Do not copy exactly to Abstract.
Budget • Give details for each category.	**References**
Curriculum vitae • Check requirements on how long CV should be.	**Acknowledgements** • Acknowledge people who made a large contribution to the paper. • Acknowledge the funding agency.
Prior support • This may or may not be requested, but can be useful if you have a good record with same funding agency.	
Other supporting documents • Include relevant documents, e.g. letters of support. • Check with Call for Proposals.	

3.8 To decide where to publish reports, researchers might consider the following:

- Journal where readership is high (There are a lot of readers.); and
- Journal with page charges vs. journal with no fee to publish.

4. Outcomes of the Workshop

4.1 Throughout the workshop, the participants obtained knowledge on how to compose proposals and reports effectively.

4.2 The participants gained practical skills to compose abstracts, following the four questions to be addressed in drafting abstracts, through the workshop's writing exercise.

4.3 Teaching materials, including lecture notes and references, were prepared and are available for future use by similar training programmes under the Yellow Sea Project as well as other environmental projects.

Annex I

List of Participants

People's Republic of China

Mr. Cui Fujun
Deputy Director
Science and Technology Division
Ocean University of China
5 Yushan Road, Qingdao
China
Tel: 86-532-82032610
Fax: 86-532-82032610
Email: fujuncui@ouc.edu.cn

Mr. Dong Jiahe
Programme officer
Division of Environmental Protection,
Marine and Fishery Bureau of Jiangsu
Province
90 New Mofan Road, Nanjing, China
Tel: 86-25-83581285
Fax: 86-25-83581288
Email: dongjiahe88180@sina.com

Mr. Han Yong
Deputy Director
Division of Environmental Protection
Marine and Fishery Bureau of Liaoning
Province
2 Taiyuan North Street, Heping District,
Shenyang, China
Tel: 86-24-23448518
Fax: 86-24-23448511
Email: hanyong@iit.edu

Mr. Li Zhaoxin
Programme officer
Division of Environmental Protection
Marine and Fishery Bureau of Shandong
Province
162 jiefang Road, Jinan, China
Tel: 86-531-86975049
Fax: 86-531-86973934
Email: lizhaoxin@hssd.gov.cn

Mr. Li Zhongyi
Research Assistant
Yellow Sea Fisheries Research Institute
106 Nanjing Road, Qingdao
China
Tel: 86-130-61495082
Fax: 86-532-85849430
Email: zhoyil@126.com

Mr. Lin Xinzhen
Senior Engineer
National Marine Environmental Monitoring
Center, SOA
42 Linghe Street, Shahe District, Dalian
116023, China
Tel: 86-411-84783299
Fax: 86-411-84783277
Email: xzlin@nmemc.gov.cn

Mr. Liu Hongjun
Director
Institute of Mariculture of Shandong Province
47 guizhou road, Qingdao
China
Tel: 86-13305320223
Fax: 86-532-82675569
Email: hongjunl@126.com

Mr. Pu Xinming
Research Assistant
First Institute of Oceanography, SOA
6 Xianxialing Road, Hi-Tech Industrial Park,
Qingdao, China
Tel: 86-532-88967447
Fax: 86-532-88967447
Email: xmpu@fio.org.cn

Ms. Zhang Kun
Assistant Editor
China Ocean News
1 Fuxingmenwai Avenue, Beijing
China
Tel: 86-10-68048051
Fax: 86-10-68048051
Email: juanjuan630@sohu.com

Ms. Zheng Wei
Doctor candidate
Ocean University of China
5 Yushan Road, Qingdao
China
Tel: 86-10-68019791
Fax: 86-10-68048051
Email: susan0537@163.com

Republic of Korea

Mr. Chang Hak-Bong
Head
Marine Environment Policy Research Team
Marine Environment & Coastal Management
Research Division
Korea Maritime Institute (KMI)
1027-4, Bangbae 3-dong, Seocho-gu
Seoul 137-851
Tel: 82-2-2105-2763
Fax: 82-2-2105-2779
Email: hbchang@kmi.re.kr

Mr. Cho Sam-Kwang
Scientist
West Sea Fisheries Research &
Development Institute, NFRDI.
707 Eulwang-dong, Jung-gu,
Incheon 400-420, ROK
Tel: 82-32-745-0571
Fax: 82- 32-745-0569
Email: skcho@momaf.go.kr

Mr. Lee Gusung
Researcher
Korea Maritime Institute (KMI)
1027-4, Bangbae 3-dong, Seocho-gu
Seoul 137-851
Tel: 82-2-2105--2956
Fax: 82-
Email: jessie@kmi.re.kr

Ms. Song Mi-Young
Fisheries Oceanographer
Fisheries resource team
West Sea Fisheries Research &
Development Institute, NFRDI.
707 Eulwang-dong, Jung-gu,
Incheon 400-420, ROK
Tel: 82-32-745-0564
Fax: 82-32-745-0569
Email: miysong@momaf.go.kr

Korea Ocean Research and Development Institute

Ms. Choi Hyo-Jin
Researcher
International Cooperations Division
Korea Ocean Research and Development
Institute (KORDI)
1270 Sa-dong Sangnok-gu Ansan-si
Gyeonggi-do 426-744
Tel: 82-31- 400-7755
Fax: 82-31- 406-6925
Email: choihj@kordi.re.kr

Ms. Choi Young-Rae
Researcher
Policy Research Division
Korea Ocean Research and Development
Institute (KORDI)
1270 Sa-dong Sangnok-gu Ansan-si
Gyeonggi-do 426-744
Tel: 82-31- 400-7757
Fax: 82-31- 406-6925
Email: yrchoi@kordi.re.kr

Mr. Kim Chang-Shik
Principal Researcher
Coastal Engineering Research Dept.
Korea Ocean Research and Development Institute (KORDI)
1270 Sa-dong Sangnok-gu Ansan-si
Gyeonggi-do 426-744
Tel: 82-31- 400-6340
Fax: 82-31- 400-6340
Email: surfkim@kordi.re.kr

Mr. Lim Hak-Soo
Researcher
Coastal Engineering Research Dept.
Korea Ocean Research and Development Institute (KORDI)
1270 Sa-dong Sangnok-gu Ansan-si
Gyeonggi-do 426-744
Tel: 82-31- 400-6334
Fax: 82-31- 400-6340
Email: hslim@kordi.re.kr

Lecturer

Mr. Christopher B. Craft
Associate Professor
Indiana University, School of Public and Environmental Affairs
1315 East 10th Street, Bloomington, IN 47405
USA
Tel: 812-855-5971
Fax: 812-855-7802
Email: ccraft@indiana.edu

Project Management Office (PMO)

Mr. Jiang Yihang
Project Manager
UNDP/GEF Yellow Sea Project
Korea Ocean Research and Development Institute
1270 Sa-dong, Sangnok
Ansan City, Gyeonggi Province 426-744
Republic of Korea
Tel: 82-31-400-7825
Fax: 82-31-400-7826
Email: yihang@yslme.org

Mr. Endo Isao
Environmental Economics Officer
UNDP/GEF Yellow Sea Project
Korea Ocean Research and Development Institute
1270 Sa-dong, Sangnok
Ansan City, Gyeonggi Province 426-744
Republic of Korea
Tel: 82-31-400-7793
Fax: 82-31-400-7826
Email: isao@yslme.org

Ms. Kim Junghwa
Secretary
UNDP/GEF Yellow Sea Project
Korea Ocean Research and Development Institute
1270 Sa-dong, Sangnok
Ansan City, Gyeonggi Province 426-744
Republic of Korea
Tel: 82-31-400-7829
Fax: 82-31-400-7826
Email: junghwa@yslme.org

Annex II

Lecture Materials

Proposal and Report Writing Workshop for Environmental Practitioners: Keys to Effective Writing

Professor Christopher Craft
Ansan ROK October 22-23, 2007

The workshop will cover how to write effective proposals and research papers, including abstracts. Lectures will described tips for writing proposals, including proposal organization, experimental design and statistics, and presentation of data (figures and tables). A hands-on exercise will illustrate how to present data as figures and tables and how to organize and list references (in-text citations and bibliographic list).

Tips for writing a research paper, including the structure of the paper, composing the abstract and use of SI (Systeme International d'Units) units, will be covered. A writing exercise to compose abstracts for presentations, proposals and research papers will be conducted.

To get the most out of the Workshop, participants are requested to bring data they have collected and analyzed statistically, and a partially completed proposal, research paper or abstract to work and improve upon.

Biographical Sketch

Professor Christopher Craft

Professor Christopher Craft is Associate Professor in the School of Public and Environmental Affairs at Indiana University Bloomington (IUB) where he conducts research, service and teaching of freshwater and estuarine wetlands ecology, restoration and management.

Professor Craft's educational background consists of degrees from North Carolina State University (PhD, Soil Science), the University of Tennessee (MS, Ecology) and the University of North Carolina (BA, Biology). His research and service programs are funded through grants from federal (U.S. National Science Foundation, U.S. Environmental Protection Agency, National Oceanic and Atmospheric Administration, U.S. Fish and Wildlife Service) and regional (Everglades National Park, North Carolina Department of Transportation, New Jersey Dept of Environmental Protection) governmental agencies and NGO's (The Nature Conservancy).

Professor Craft is President-elect of the *Society of Wetland Scientists* (2007-2008), a 3500 member international organization, and past-president of Division S-10, Wetland Soils, of the *Soil Science Society of America*.

At IUB, Professor Craft teaches courses in Wetlands Ecology, Restoration Ecology, Applied Ecology and Environmental Science. Professor Craft advises undergraduate, Master's and PhD student research. Professor Craft and his students have published more than 60 peer-reviewed research papers and given more than 100 presentations at national and international meetings.

Proposal and Technical Writing Workshop UNDP YSLME

Ansan, South Korea
October 22-23, 2007

Prof. Christopher Craft
Indiana University
School of Public and Environmental Affairs

Proposal Writing

Before You Begin

- Know your funding agency and their needs
- Funding Agencies
  - National Science Foundation (NSF)
  - Environmental Protection Agency (EPA)
  - United Nations Environmental Protection (UNEP)
  - Other regional or state agencies, and NGO's

Funding Announcements

- **Two types**:
  - RFP – request for proposals
  - RFA – request for assistance
- **Read these CAREFULLY**
- **Identify Agency's priorities and needs**
- **Tailor your proposal to address as many of their priorities and needs as possible.**
- **Follow the proposal instructions to the letter**

Formatting your Proposal

- **Check with the Funding agency for any specific requirements**
- **A Typical Proposal contains:**
  - I. Abstract
  - II. Introduction
  - III. Hypothesis
  - IV. Method
  - V. Relevance/Significance to the RFP
  - VI. Synergies/Linkages with other projects
  - VII. References
  - VIII. Boilerplate Materials

I. Abstract

- **One-page summary of:**
  - **What you are proposing**
  - **Why you are proposing it *(i.e. what questions you will answer or what hypothesis you will test)***
  - **How you will test it *(i.e. what methods you will use)***
  - **Why it is important *(i.e. why should the agency fund your proposal instead of someone else's).***
- **The Abstract should be the LAST section you write.**

II. Introduction

- **Explain the nature of the problem or question (2-3 pages)**
  - What is known about the problem?
  - What is not known?
  - What are the information gaps?
- **Use supporting or preliminary data to support your hypothesis**
- **Supporting data are VERY important**
  - Use figures or tables
  - Figures are easier to interpret

III. Hypothesis

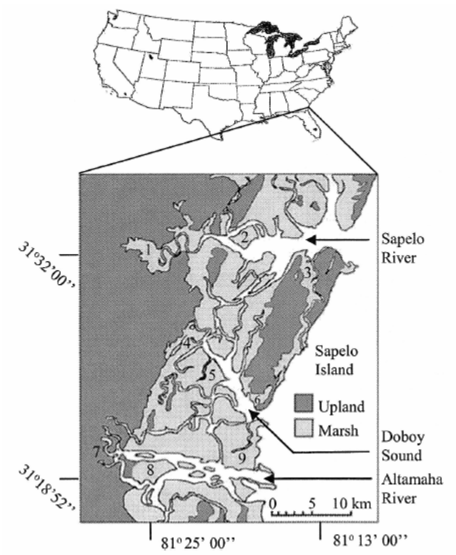
- **Briefly state your research question(s)**
- **A conceptual diagram illustrating the hypothesis is helpful**
- **Your hypothesis should be formulated first, before any other part of the proposal**
- **The Hypothesis is the Most Important Part of the Proposal!**

IV. Methods

- This is the longest section and should include:

- Site Description
  - Map
  - Detail the environmental conditions
    - General climate
    - Soils
    - Vegetation
    - Water currents
    - Water chemistry

Importance of a Site Map



IV. Methods

- Experimental Design
  - Sampling protocol (temporal/spatial) – how frequently and in what manner will you sample across the study sites?
  - Replication of sample collection
  - Sampling scheme
    - Randomly sampled using random number generator to select locations.

- Experimental Design is the second most important section, after the hypothesis.

Experimental Design

- When different treatments are used in research, two designs—treatment design and experimental design—are needed.

- Treatment Design describes controlled levels of factors such as temperature, salinity or nutrient additions, different genotypes or species, different soil types.

- Experimental Design describes the method of arranging the experimental units and the method of assigning treatments.

Experimental Design

- **Included should be:**
  - **the number of replicates,**
  - **a description of conditions at the field sites or in the greenhouse or lab,**
  - **the number of sites or years that measurements are made,**
  - **what measurements are made and how they are made.**

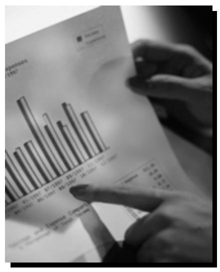
IV. Methods

- **Field Sampling**
  - **How samples will be physically collected**
  - **How much of a sample will be collected**
  - **What testing is done in the field**
  - **How will samples be transported and preserved**

IV. Methods

- Laboratory Analysis
  - What constituents will you measure and with what methods
  - Always cite/reference widely accepted methods that are published in peer-reviewed sources (e.g. Methods text books, methods articles in journals)

IV. Methods



- Statistical Methods
  - Dependent upon experimental design
  - <u>Experiments</u>: analysis of variance (ANOVA), repeated measures ANOVA or regression to investigate the effects of controlled treatments
  - <u>Monitoring</u>: correlation analysis, regression or multivariate analysis
  - <u>Non-Parametric (Rank) test</u>: use if the data do not meet the assumptions required of the statistical methods

Abbreviations used in Statistics

Table 4–1. Some widely used abbreviations and symbols in statistics.†

Name	Abbreviation or symbol‡
Abbreviations	
Analysis of variance	ANOVA
Coefficient of variation	CV
Completely randomized design	CRD
Degrees of freedom	df
Fisher's least significant difference	LSD
Multivariate analysis of variance	MANOVA
Mean square error	MSE
Nonsignificant (or not significant)	NS
Randomized complete block design	RCB or RCBD
Root mean square error	RMSE
Standard deviation	SD
Standard error	SE
Standard error of the mean	SEM

Symbols used in Statistics

Symbols	
Arithmetic mean	$\bar{x}$ (μ)
Chi-square statistic	χ^2
Correlation coefficient	r (ρ)
Coefficient of multiple determination	R^2
Probability of a Type I error	α
Probability of a Type II error	β
p value	p
Regression coefficient	b (β)
Sample size	n
Standard deviation	s (σ)
Standard error of the mean	$s_{\bar{x}}$ ($\sigma_{\bar{x}}$)
Student's t statistic	t
Snedecor's F statistic	F
Variance	s^2 (σ^2)

† In addition, the symbols *, **, and *** are used to show significance at the α = 0.05, 0.01, and 0.001 levels, respectively. Significance at other levels is designated by additional footnotes, using the next available symbol from the standard sequence (†, ‡, §, ¶, #, ††, ‡‡, etc.).

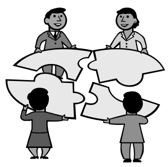
‡ Symbols in parentheses are for the population analog of the corresponding sample quantity.

V. Relevance/ Significance to RFP

- Discuss how your proposal addresses the priorities of the RFP
- How will your work answer key questions or information gaps?
- What is novel about the proposed work?

VI. Synergies/Linkages with Other projects

- It is important to demonstrate how your proposal can leverage additional resource to "add value" to the proposed work
- Are there other studies in the area whose data you can use to increase the value of your work?
- Look for available monitoring data to augment/supplement your data
- Access to infrastructure (ship time, lab facilities) may enable you to collect/obtain more data without spending additional money for it.



VII. References

- A compilation of the peer-reviewed literature that you cite in the proposal text.
- Check for formatting specifications in the RFP

References

- A complete list of the peer-reviewed and other literature cited in the manuscript
- Arrange the list alphabetically. Two or more papers by the same author(s) are arranged chronologically
- Examples: Journal articles, books, book equivalents-bulletin, symposium or proceedings, chapter in a book, chapter in a symposium or proceedings, dissertation or thesis, software and software documentation.

References

- In the text, cite references by name and year (i.e. Smith 1997, Jones and Smith 1998, Smith et al. 2000).
- Arrange the list alphabetically by the surnames of authors.
- Two or more articles by the same author (or authors) are listed chronologically;
- Two or more articles with the same in-text citation are indicated by the letters a,b,c etc.

References

- All single-authored articles of a given individual should precede multiple-author articles of which the individual is senior author.
- Alphabetize entries with the same first author according to surnames of succeeding coauthors and then by year, when the names are repeated exactly.

Citing References in the Text: An Example

Soils are an important component of tidal marshes. They sequester organic matter, N, and P (Craft et al. 1988), support complex biogeochemical reactions (Capone and Kiene 1988), and contribute to long-term marsh stability through deposition of mineral sediment and accumulation of organic matter (DeLaune et al. 1983; Hatton et al. 1983; Nyman et al. 1990; Craft et al. 1993; Morris et al. 2002).

Listing References in the Bibliography: An Example

CAPONE, D. G., AND R. P. KIENE. 1988. Comparative microbial dynamics in marine and freshwater sediments: Contrasts in anaerobic carbon catabolism. Limnol. Oceanogr. **33:** 725–749.

COMANS, R. N. J., J. J. MIDDLEBURG, J. ZONDERHUIS, J. R. W. WOITTIEZ, G. J. DE LANGE, H. A. DAS, AND C. H. VAN DER WEIJDEN. 1989. Mobilization of radiocaesium in pore water of lake sediments. Nature **339:** 367–369.

CONN, C. E., AND F. P. DAY, JR. 1997. Root decomposition across a barrier island chronosequence: Litter quality and environmental controls. Plant Soil **195:** 351–365.

CRAFT, C. B., S. W. BROOME, AND E. D. SENECA. 1988. Nitrogen, phosphorus and organic carbon pools in natural and transplanted marsh soils. Estuaries **11:** 272–280.

———, ———, AND ———. 1991. Porewater chemistry of natural and created marsh soils. J. Exp. Mar. Biol. Ecol. **152:** 187–200.

References

Shotwell, O.L. 1998.
Shotwell, C.A., and G.W. Smith. 2001.
Shotwell, O.L., M.L. Goulden, and C.W. Hesseltine. 1994.
Shotwell, O.L., C.W. Hesseltine, and M.L. Goulden. 1993a.
Shotwell, O.L., C.W. Hesseltine, and M.L. Goulden. 1997.
Shotwell, O.L., C.W. Hesseltine, E.E. Vandegraft, and M.L. Goulden. 1993b.
Shotwell, O.L., W.F. Kwolek, M.L. Goulden, L.K. Jackson, and C.W. Hesseltine. 1991.
Shotwell, O.L., and D.W. Zweig. 1994.

VIII. Boilerplate Materials

- This will be detailed in the RFP
- Usually need a CV (1-2 pages)
- Current and pending support from other research projects
- A description of field and lab facilities available for use
- Letters of support from partners with whom you have linkages

Other Considerations

1. The Nature of the funding agency

- For example: NSF vs. EPA
- NSF
  - Funds proposals addressing basic science questions that advance the field.
  - Tends to be hypothesis-driven, experimental and manipulative in nature
- EPA
  - Funds monitoring studies that are less experimental and observational in nature
  - Tends to fund broader, multidisciplinary, multi-investigator studies

Other Considerations

2. Importance of Citing Peer-reviewed Literature

- This is especially important in the Introduction
- Lays the groundwork for what is known about the question/problem.
- Hypothesis should address what is NOT KNOWN and, thus needs to be funded.
- A proposal that reinvents the wheel will not be funded

Other Considerations

3. Single vs. Multi-Investigator Proposals

- Depends on the scope (and amount of money) in the RFP
- Partner with researchers who bring other skills to build a team
- Skills may range, and team members will vary depending on project goals:
  - Micro- (molecular ecologists) to macro- (GIS) scales
  - Different research expertise or different media (air, water, soils)

Other Considerations

3. Single vs. Multi-Investigator Proposals

- In addition to data collection and analysis, a synthetic approach such as simulation modeling may strengthen the proposal.
- Team members may come from entirely different disciplines (e.g. social scientists, economists) to better link science, policy and public interest.
- Stakeholder involvement (e.g. community groups) may be important to consider
- Agencies increasingly fund larger, better financed proposals that require such a multi-disciplinary approach

Use of Tables and Figures

- Tables and figures are an integral part of a well written paper.
- Tables enable you to present a lot of comparable data in a small amount of space.
- Figures enable you to show trends or patterns in the data.
- As you prepare to present your information, think over whether a table, figure or text is more appropriate

Use of Tables and Figures

- If the text is crowded with detail, especially comparable quantitative data, consider creating a table
- Do not overload the text with information that could be presented better in a table
- Consolidate similar information into one table, to let the reader compare easily.

Use of Tables and Figures

- **Do not make the reader search for information**
- **If a table has only a few rows and columns, try stating the findings in a few sentences.**
- **Do not use too many tiny tables for information that could be presented better in the text.**

Use of Tables and Figures

Table 4. Correlations of soil properties with salinity for tidal marshes of Georgia and the conterminous United States.

	Georgia	Conterminous U.S.
Soil properties		
Bulk density (g cm^{-3})	0.25	0.63***
Organic C (%)	−0.70**	−0.53**
Organic C (mg cm^{-3})	−0.70**	−0.11
Nitrogen (%)	−0.73***	−0.49*
Phosphorus ($\mu g\ g^{-1}$)	0.03	−0.49*
C : N (mol : mol)	−0.16	0.05
N : P (mol : mol)	−0.63	−0.07
Soil accumulation		
Feldspar accretion (mm yr^{-1})	−0.94**	—
^{137}Cs accretion (mm yr^{-1})	−0.69**	−0.47*
Sediment (g m^{-2} yr^{-1})	ns	−0.06
Organic C (g m^{-2} yr^{-1})	−0.79***	−0.27
Nitrogen (g m^{-2} yr^{-1})	−0.79***	−0.57*
Phosphorus (g m^{-2} yr^{-1})	−0.61**	−0.05

ns, Not significant.
* $p<0.05$.
** $p<0.01$.
*** $p<0.001$.

Use of Tables and Figures

Table 3. Soil bulk density, organic C, total N, total P, C:N, and N:P (0–30 cm depth) of freshwater-dominated (Altamaha River) versus marine-dominated marshes (Doboy Sound, Sapelo River).*

		Bulk density (g cm^{-3})		Organic C (%)		Nitrogen (%)		Phosphorus (μg g^{-1})		C:N (mol:mol)		N:P (mol:mol)	
Site No.	Estuary	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain
1	Sapelo River	0.23	1.09	11.3	1.4	0.51	0.08	450	260	26	20	25	7
2		0.47	0.43	4.1	4.2	0.23	0.25	580	670	21	20	9	8
3		0.39	0.41	5.1	4.2	0.35	0.30	860	590	17	16	9	11
4	Doboy Sound	1.10	1.03	1.4	2.1	0.09	0.13	100	80	17	19	20	35
5		0.30	0.35	5.5	6.6	0.37	0.36	450	600	18	21	18	13
6		0.41	0.41	5.2	4.6	0.34	0.32	520	520	18	17	15	13
7	Altamaha River	0.32	0.19	8.0	14.7	0.54	0.93	560	520	17	18	21	39
8		0.31	0.22	10.1	16.2	0.65	0.84	640	560	18	23	22	33
9		0.27	0.38	7.9	4.1	0.50	0.40	510	648	19	18	21	9
Mean Sapelo (n = 6)		0.50±0.12 a		5.0±1.4 a		0.29±0.06 a		570±80 a		20±1 a		11±3 a	
Mean Doboy (n = 6)		0.60±0.15 a		4.2±0.8 a		0.27±0.05 a		380±90 a		18±1 a		19±3 ab	
Mean Altamaha (n = 6)		0.28±0.03 a		10.2±1.8 b		0.64±0.08 b		570±20 a		18±1 a		25±4 b	

* Estuary means separated by the same letter are not significantly different (α<0.05) according to the Ryan-Einot-Gabriel-Welsch multiple range test.

Use of Tables and Figures

- **In a difficult prose (written) explanation, ask yourself if an illustration would help.**
- **Do not struggle to describe something in words that could be shown much more easily in a figure**
- **Look at your figures, do they show more than could be said in a few well-chosen words?**
- **Do not assume that a picture is always better.**

Use of Tables and Figures

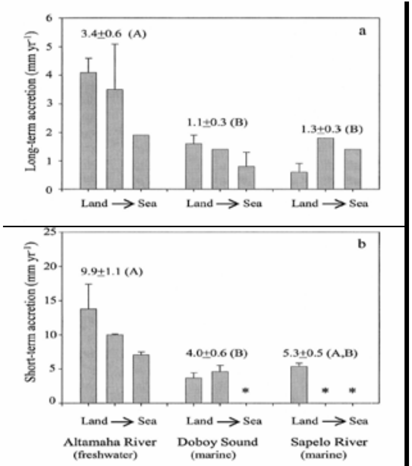


Fig. 2. (a) Long-term (^{137}Cs) and (b) short-term (feldspar marker) vertical accretion of freshwater- (Altamaha River) and marine- (Doboy Sound, Sapelo River) dominated marshes along the Georgia coast. Means ± 1 SE are shown for each estuary. Means separated by the same letter (in parentheses) are not significantly different ($p \leq 0.05$) on the basis of the Ryan-Einot-Gabriel-Welsch multiple range test. Locations marked with an asterisk (*) were omitted: the feldspar marker layer could not be detected because of bioturbation by fiddler crabs.

Use of Tables and Figures

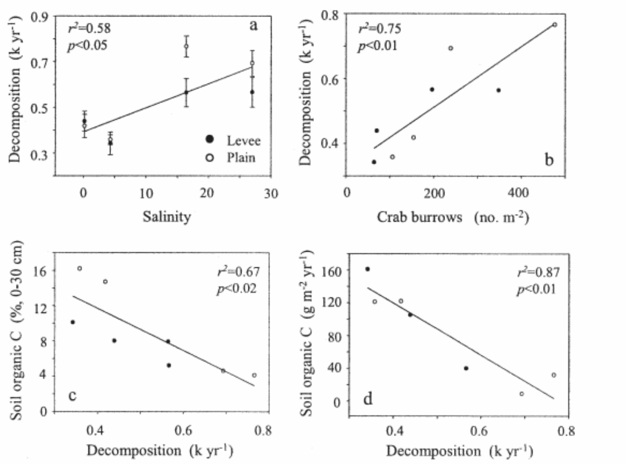


Fig. 4. Relationships between decomposition rate and (a) salinity and (b) crab burrow density, and (c) percent soil organic C and (d) organic C accumulation versus decomposition rate in four tidal marshes exposed to different salinities. Salinity is based on daily (2002 and 2003) measurements in the water column of the adjacent estuary. Burrow density is based on spring and fall 2003 counts of crab burrows (n = 4) per marsh zone per season (Dale Bishop unpubl. data).

Abstracts

- Abstracts are used to summarize the key points of a study, proposal, oral presentation, or poster.
- Abstracts are required for proposals and journal articles.
- They are often required for presentations (oral and poster) at scientific meetings and conferences.
- Abstracts usually are NO MORE than one page long.

Writing the Abstract

- **What did you do?...**
  - **"We measured bulk density, nutrients (carbon-C, nitrogen-N, phosphorus-P), accretion, and accumulation were compared in tidal marshes of three estuaries of Georgia that varied in delivery of freshwater."**
- **Why do you do it?...**
  - **"to identify relationships between freshwater input and marsh soil properties."**

Writing the Abstract

- **What did you find?...**
  - **"Soil organic carbon (C) and nitrogen (N) (0-30cm) were two times greater in marshes of the freshwater-dominated Altamaha River than in the salt marshes of Doboy Sound and Sapelo River. 137Cs accretion and accumulation of organic C and N were three to five times greater in freshwater-dominated marshes than in salt marshes. The patterns observed in Georgia marshes were geographically general; data for tidal freshwater and brackish marsh soils compiled from 61 studies in the conterminous United States (U.S.) showed lower bulk density and higher percent organic C and N than salt marshes, regardless of geographic region."**

Writing the Abstract

- **What did you find?...**
  - **"Salinity, a proxy for freshwater input, was inversely correlated with percent soil organic C, N and vertical accretion in Georgia marshes and in marshes elsewhere in the conterminous U.S. There was no relationship between above- and below-ground emergent plant production and salinity of Georgia marshes but the rate of root decomposition was positively related to salinity, and decomposition rate was negatively related to percent soil organic C and C accumulation."**

Writing the Abstract

- **Why is it important?...**
  - **"In Georgia tidal marshes and elsewhere, soil organic matter content and accumulation are mediated by freshwater through its effects on decomposition. My findings suggest that accelerated sea level rise and saltwater intrusion caused by climate change may increase decomposition of soil organic matter that may reduce vertical accretion and adversely affect long-term marsh stability."**

Abstract

- **Examples of different Abstract formats:**
  - **1. Abstract submitted to a conference (Wetland Biogeochemistry)**
  - **2. Abstract as a part of a journal paper (Limnology and Oceanography)**
  - **3. Abstract as part of a proposal (Department of Energy Proposal)**

Introduction

- **Several paragraphs to a page or two explaining:**
  - **The nature of the problem or question (i.e. what is known about the problem)**
  - **What is not known,**
  - **What are the information gaps?**
- **The introduction is supported by extensive peer-reviewed literature.**
- **You state the hypothesis, objectives or questions at the end of this section.**

Methods

- Be sure to include:
  - Site Description
  - Field Sampling
  - Laboratory Analysis
  - Statistical Methods

- More details in the “Proposal Writing” notes.

Results and Discussion

- Results:
  - Present your findings
  - Use graphs and tables

- Discussion:
  - Discuss how your results relate to your hypotheses
  - How does your results compare to other published studies?

Discussion

- Do your findings support your hypothesis? Why?...or Why not?
- How do your results compare with other published studies? Did you find similar results?...or not? Why?
- Do your findings pose new (unanswered) questions that are important and, potentially “break new ground” in your field?

Conclusions

- May be a section or a paragraph at the end of the manuscript
- It briefly states the most important findings, their significance to the field and any new questions that are posed.
- It is shorter than the Abstract

Example of a Conclusions

In Georgia tidal marshes and elsewhere, freshwater input promotes organic matter preservation and accumulation. In Georgia, short- and long-term accretion, percent soil organic C, N and N : P, and accumulation of organic C and N were greater in tidal marshes of the freshwater-dominated Altamaha River than in salt marshes of Doboy Sound and Sapelo River. In situ decomposition of roots was greater in salt marshes than in the tidal freshwater and brackish marshes and it was positively related to surface water salinity. Percent soil organic C and organic C accumulation were inversely related to decomposition but were unrelated to above- or belowground emergent plant production.

Example of a Conclusions (continued)

Freshwater-driven, landscape-scale patterns of soil properties observed in Georgia tidal marshes also occur in other geographic regions of the conterminous United States. In a survey of 61 published and two unpublished studies, bulk density was lower and percent organic C, N, and P were consistently greater in tidal freshwater marshes and brackish marshes than in salt marshes regardless of geographic region. These findings suggest that freshwater input is important in structuring tidal marsh soils across a wide range of climatic and geomorphic conditions because of its association with lower decomposition rates relative to areas with greater seawater influence.

Other Sections

- References (already covered)
- Acknowledgements

Acknowledgements

- May come at the beginning or end of the manuscript.
- Acknowledge the people (Not co-authors) who measurably contributed to the success of work.
- This might include people who helped with field sampling, lab analysis and technical support.
- Be sure to acknowledge and thank your funding source(s) here.

Acknowledgements

Acknowledgments

The idea for this work grew out of two National Science Foundation (NSF)–sponsored workshops, *Regulation of Organic Matter Preservation in Wetland Sediments* held at NSF Long Term Ecological Research (LTER) All Scientists Meeting, Snowbird, Utah (1–5 August 2000) and *Soil Organic Matter in Wetlands* held at Virginia Institute of Marine Sciences, Gloucester Point VA (26–29 July 2001). Thanks to Ken Helm, Sean Graham, and Josh Hall for their assistance in the field; Chrissy Pruett, Jillian Bertram, and Sarah Butler for preparing and analyzing soil samples; and Steve Pennings, Dale Bishop, and Samantha Joye for sharing data. Steve Pennings, Pat Megonigal, Ross Brittain, and two anonymous reviewers provided constructive comments on previous versions of the manuscript.

This work was supported by grant OCE-9982133 from the National Science Foundation and is a contribution of the Georgia Coastal Ecosystems LTER program.

This is Contribution 889 of the University of Georgia Marine Institute.

SI (System International d’Units)

- Required for publications in peer-reviewed journals
- Based on seven base units
- Derived units, such as Celsius (C), are expressed algebraically as base units

Base Units

Table 7–1. Base

Quantity	Symbol	Unit
Length	m	meter
Mass	kg	kilogram
Time	s	second
Electric current	A	ampere
Thermodynamic	K	kelvin
Amount of subs	mol	mole
Luminous inten	cd	candela

Derived Units

Name	Symbol	Expression in Terms of SI Units
Celsius Temperature	C	K
Pressure (pascal)	Pa	kg s^{-1}
Electrical conductance (siemens)	S	m^{-2} kg^{-1} s^{3} A^{2}
Energy, work, quantity of heat (joule)	J	m^{2} kg s^{-2}
Activity of a radionuclide (becquerel)	Bq	s^{-1}

Prefixes

Order of magnitude	Prefix	Symbol
10^{24}	yotta	Y
10^{21}	zetta	Z
10^{18}	exa	E
10^{15}	peta	P
10^{12}	tera	T
10^{9}	giga	G
10^{6}	mega	M
10^{3}	kilo	k
10^{2}	hecto†	h†
10^{1}	deka‡	da‡

† To be avoided when possible.
‡ Not to be used.

Prefixes

Order of magnitude	Prefix	Symbol
10^{-1}	deci†	d†
10^{-2}	centi†	c†
10^{-3}	milli	m
10^{-6}	micro	μ
10^{-9}	nano	n
10^{-12}	pico	p
10^{-15}	femto	f
10^{-18}	atto	a
10^{-21}	zepto	z
10^{-24}	yocto	y

Non-SI Units

- ❑ **Sometimes Non-SI units may be used in publications**

<u>Temperature:</u> **^{o}C rather than ^{o}K**

<u>Area:</u> **Hectares rather than m^2**

<u>Volume:</u> **Liters rather than m^3**

Reporting Concentrations

- **Express concentrations on a molar basis:**

$$1 \text{ mol L}^{-1} = 1\ M = 1 \text{ mmol mL}^{-1}$$
$$1 \text{ mmol L}^{-1} = 1 \text{ m}M = 10^{-3}\ M = 1\ \mu\text{mol mL}^{-1}$$
$$1\ \mu\text{mol L}^{-1} = 1\ \mu M = 10^{-6}\ M = 1 \text{ nmol mL}^{-1}$$
$$1 \text{ nmol L}^{-1} = 1 \text{ n}M = 10^{-9}\ M = 1 \text{ pmol mL}^{-1}$$

Preferred vs. Acceptable Units

- **Units such as percentages, parts per thousand, and parts per million are ambiguous.**
- **Where ever possible, data should be expressed in SI units.**
- **Parts Per thousand: Used for reporting stable isotope values relative to a standard (expressed as parts per thousand or per mil, $^{o}/_{oo}$)**

Preferred vs. Acceptable Units

- Percentage (%) is acceptable in the following cases:
  - Coefficient of variation (in statistics)
  - Botanical composition such as percent coverage (or cover)
  - Relative humidity

- Parts Per Million is Not recommended

Preferred vs. Acceptable Units

Quantity	Application	Unit	Symbol
Concentration	known molar mass (liquid or solid)	mole per cubic meter (P)	$mol\ m^{-3}$
		mole per kilogram (P)	$mol\ kg^{-1}$
		mole per liter (A)	$mol\ L^{-1}$
		gram per liter (A)	$g\ L^{-1}$
	unknown molar mass (liquid or solid)	gram per cubic meter (P)	$g\ m^{-3}$
		gram per kilogram (P)	$g\ kg^{-1}$
		gram per liter (A)	$g\ L^{-1}$
	known ionic charge	mole charge per cubic meter (P)	$mol_c\ m^{-3}$
		mole charge per liter (A)	$mol_c\ L^{-1}$
	gas	mole per cubic meter (P)	$mol\ m^{-3}$
		gram per cubic meter (A)	$g\ m^{-3}$
		gram per liter (A)	$g\ L^{-1}$
		liter per liter (A)	$L\ L^{-1}$
		microliter per liter (A)	$\mu L\ L^{-1}$
		mole per liter (A)	$mol\ L^{-1}$
		mole fraction (A)	$mol\ mol^{-1}$
Light	irradiance	watt per square meter	$W\ m^{-2}$
	photosynthetic photon flux density (400–700 nm)	micromole per square meter per second	$\mu mol\ m^{-2}\ s^{-1}$

Conversion Factors

To convert Column 1 into Column 2, multiply by	Column 1 SI Unit	Column 2 non-SI Units	To convert Column 2 into Column 1, multiply by
		Length	
0.621	kilometer, km (10^3 m)	mile, mi	1.609
1.094	meter, m	yard, yd	0.914
3.28	meter, m	foot, ft	0.304
1.0	micrometer, μm (10^{-6} m)	micron, μ	1.0
3.94×10^{-2}	millimeter, mm (10^{-3} m)	inch, in	25.4
10	nanometer, nm (10^{-9} m)	Angstrom, Å	0.1
		Area	
2.47	hectare, ha	acre	0.405
247	square kilometer, km^2 (10^3 m)^2	acre	4.05×10^{-3}
0.386	square kilometer, km^2 (10^3 m)^2	square mile, mi^2	2.590
2.47×10^{-4}	square meter, m^2	acre	4.05×10^3
10.76	square meter, m^2	square foot, ft^2	9.29×10^{-2}
1.55×10^{-3}	square millimeter, mm^2 (10^{-3} m)^2	square inch, in^2	645

Conversion Factors

		Volume	
9.73×10^{-3}	cubic meter, m^3	acre-inch	102.8
35.3	cubic meter, m^3	cubic foot, ft^3	2.83×10^{-2}
6.10×10^4	cubic meter, m^3	cubic inch, in^3	1.64×10^{-5}
2.84×10^{-2}	liter, L (10^{-3} m^3)	bushel, bu	35.24
1.057	liter, L (10^{-3} m^3)	quart (liquid), qt	0.946
3.53×10^{-2}	liter, L (10^{-3} m^3)	cubic foot, ft^3	28.3
0.265	liter, L (10^{-3} m^3)	gallon	3.78
33.78	liter, L (10^{-3} m^3)	ounce (fluid), oz	2.96×10^{-2}
2.11	liter, L (10^{-3} m^3)	pint (fluid), pt	0.473
		Mass	
2.20×10^{-3}	gram, g (10^{-3} kg)	pound, lb	454
3.52×10^{-2}	gram, g (10^{-3} kg)	ounce (avdp), oz	28.4
2.205	kilogram, kg	pound, lb	0.454
0.01	kilogram, kg	quintal (metric), q	100
1.10×10^{-3}	kilogram, kg	ton (2000 lb), ton	907
1.102	megagram, Mg (tonne)	ton (U.S.), ton	0.907
1.102	tonne, t	ton (U.S.), ton	0.907

Conversion Factors

	Temperature		
1.00 (K – 273)	kelvin, K	Celsius, °C	1.00 (°C + 273)
(9/5 °C) + 32	Celsius, °C	Fahrenheit, °F	5/9 (°F – 32)
	Radioactivity		
2.7×10^{-11}	becquerel, Bq	curie, Ci	3.7×10^{10}
2.7×10^{-2}	becquerel per kilogram, Bq kg^{-1}	picocurie per gram, pCi g^{-1}	37
100	gray, Gy (absorbed dose)	rad, rd	0.01
100	sievert, Sv (equivalent dose)	rem (roentgen equivalent man)	0.01

For more information on writing style, please go to American Society of Agronomy (ASA) website…

https://www.agronomy.org/publications/style/

Environmental Monitoring

- What is Environmental Monitoring?
- Why is it important?
- What can it be used for?

Environmental Monitoring

- Environmental monitoring involves regular collection of data at fixed locations for an extended period of time.
- Monitoring data are used to identify baseline conditions and trends in environmental quality parameters, such as temperature, salinity, and nutrients.
- Monitoring data helps to show whether these trends are increasing or decreasing.

Environmental Monitoring

- Over time, these data can be used to determine whether a condition (i.e. water quality, air pollution, greenhouse gases) is getting better or worse.
- Monitoring data are collected by agencies involved in:
  - Environmental Compliance (EPA),
  - Research agencies (NSF Long-Term Ecological Research Program)
  - and sometimes Philanthropic organizations

Environmental Monitoring

- Monitoring data can be used to identify whether a corrective action (e.g. nutrient management plan to reduce nutrients), is working
- Monitoring data becomes more valuable with time because long-term (decadal) continuous data are hard to find.
- It is often difficult to get continuous financial support for long-term data collection.

Who needs environmental monitoring?

By:
Gary M Lovett, Douglas A Burns, Charles T. Driscoll,
Jennifer B. Shanley, Gene E. Likens, Richard Hauber

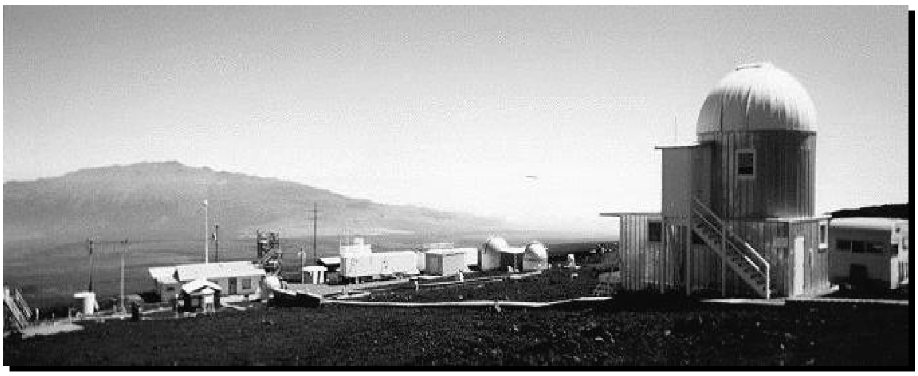
Citation: Lovett GM, DA Burns, CT Driscoll, JB Shanley, GE Likens, R Hauber. 2007. Who Needs Environmental Monitoring? Frontiers in Ecological Monitoring 5(5): 253-260

Environmental Monitoring

In a nutshell:

- Environmental monitoring is often criticized as being unscientific, expensive, and wasteful
- We argue that monitoring is a crucial part of environmental science, costs very little relative to the value of the resources it protects and the policy it informs, and has added value in that basic environmental monitoring data can be used for multiple purposes
- Effective monitoring programs address clear questions, use consistent and accepted methods to produce high-quality data, include provisions for management and accessibility of samples and data, and integrate monitoring into research programs that foster continual examination and use of the data
- Government agencies should commit to long-term support for valuable monitoring programs, and funders of basic ecological and environmental research should recognize that monitoring is a fundamental part of environmental science

Environmental Monitoring



Mauna Loa Observatory

Environmental Monitoring

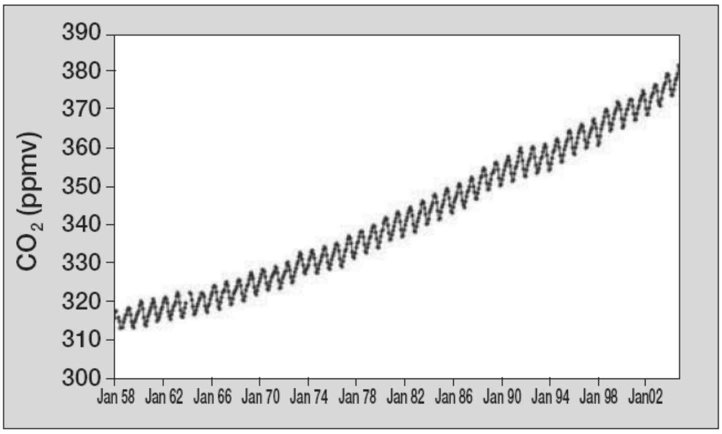


Figure 3. *Mean monthly atmospheric concentrations of CO_2 measured at Mauna Loa, HI, by CD Keeling and colleagues from 1958 to 2004. Plotted from data available at the Carbon Dioxide Information Analysis Center (www.cdiac.ornl.gov).*

Environmental Monitoring

Figure 5. *The V-notch weir at Watershed 6 of the Hubbard Brook Experimental Forest, NH. The long-term stream chemistry monitoring data shown in Figure 4 are from this stream.*

Environmental Monitoring

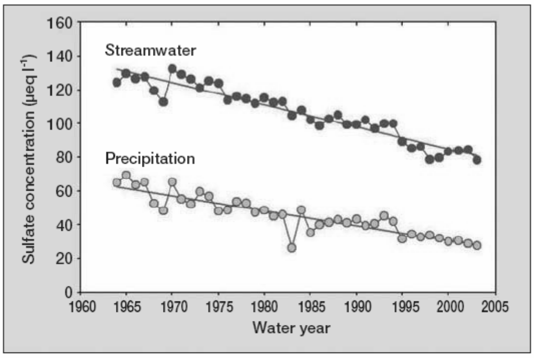


Figure 4. *Long-term record of annual volume-weighted mean sulfate concentrations in bulk precipitation and streamwater in Watershed 6 of the Hubbard Brook Experimental Forest, NH (updated from Likens et al. 2002). This data record was initiated as part of a basic study of forest ecosystem function, but has also proven valuable in assessments of ecosystem response to policies controlling emissions of air pollutants. Monitoring at the Hubbard Brook Experimental Forest relies on an effective collaboration between academic scientists and the US Department of Agriculture Forest Service, which monitors hydrological and meteorological variables at this site.*

Environmental Monitoring

Figure 6. *The lilac variety* Syringa chinensis, *"red rothomagensis", in full bloom. Flowering dates of clones of this plant are used for monitoring phenological trends associated with climate change throughout the US.*

Seven Habits of Highly Effective Environmental Monitoring Programs

Yellow Sea

Effective Monitoring

1. Design the program around clear and compelling scientific questions
2. Include review, feedback, and adaptation in the design
3. Choose measurements carefully and with the future in mind.
4. Maintain quality and consistency of the data

Effective Monitoring

5. Plan for long-term data accessibility and sample archiving
6. Continually examine, interpret, and present the monitoring data.
7. Include monitoring within an integrated research program.

Cover Page

DOE National Institute for Climatic Change Research (NICCR)

1. Project title: **Effects of Accelerated Sea Level Rise and Variable Freshwater River Discharge on Water Quality Improvement Functions of Tidal Freshwater Floodplain Forests**

2. Principal Investigator (PI) name: Christopher Craft
   Employing institution: Indiana University/SPEA
   Mailing address: 1315 E. 10th St., room 410J
   Bloomington, IN 47405
   Phone number: 812-855-5971
   Fax number: 812-855-1428
   E-mail address: ccraft@indiana.edu

3. Co-Investigator(s) (for Collaborative Projects only, as defined in the RFP; add lines as needed):
   Name, Employing institution:
   Name, Employing institution:

4. Total budget request for the project period: $343,181
   (For Collaborative Projects this is the total budget for all institutions)
   Duration of the project (years): 3

5. Certifying representative of the PI's employing institution
   Name: Steven A. Martin, Assoc. Vice Provost for Research

   Signature: Steven A. Martin 8.7.07

   Employing institution: Indiana University
   Mailing address: P.O. Box 1847
   Bloomington, IN 47402
   Phone number: 812-855-0516
   Fax number: 812-855-9943
   E-mail address: rugs@indiana.edu

6. Contractual contact of the PI's employing institution
   Name: Marcia Landen
   Employing institution: Indiana University
   Mailing address: P.O. Box 1847
   Bloomington, IN 47402
   Phone number: 812-855-0516
   Fax number: 812-855-9943
   E-mail address:

7. Date of submission: 8/21/2007

8. Has this proposal been submitted to another Federal agency? No

9. Has this proposal been submitted to another DOE program? No

DOE National Institute for Climatic Change Research: Budget Page Year One

Organization: Indiana University/SPEA
Principal Investigator: Christopher Craft
Budget Start Date: 04/01/08 Budget End Date: 05/30/09

A. Senior Personnel	Person-months	Funds requested
1. Christopher Craft	1 mo/summer	8,894
2. Craig Wayson	2 mo/summer	8,240
3.		
4.		
5.		
6.		
B. Other Personnel (list number in parentheses)		
1. (1) Post-doctoral associates	12 mos.	18,000
2. ( ) Other professional (technician, programmer, etc.)		
3. ( ) Graduate students		
4. ( ) Undergraduate students		
5. ( ) Secretarial - Clerical		
6. (1) Other		4,000
Total Salaries and Wages (A+B)		
C. Fringe Benefits (if charged as direct cost)		5,237
Total Salaries, Wages, and Fringe Benefits (A+B+C)		44,371
D. Permanent Equipment (list each item and dollar amount; insert lines if needed)		25,103
1.		
2.		
3.		
Total Permanent Equipment		
E. Travel 1. Domestic		6,000
2. Foreign		
Total Travel		6,000
F. Trainee/Participant Costs (not applicable to NICCR)		**not applicable**
G. Other Direct Costs		
1. Materials and Supplies		6,000
2. Publication Costs		
3. Consultant Services		
4. Computer (ADPE) Services		
5. Subcontracts (total for all subcontracts)		
6. Other		12,680
Total Other Direct Costs		18,680
H. Total Direct Costs (A through H)		94,154
I. Total Indirect Costs (specify rates and bases) Total direct less fee remissions 51.5%		30,576
J. Total Direct and Indirect Costs (H+I)		124,730

DOE National Institute for Climatic Change Research: Budget Page Year Two

Organization: Indiana University/SPEA
Principal Investigator: Christopher Craft
Budget Start Date: 04/01/09 Budget End Date: 05/30/10

A. Senior Personnel	Person-months	Funds requested
1. Christopher Craft	1 mo/summer	9,161
2. Craig Wayson	2 mo/summer	8,487
3.		
4.		
5.		
6.		
B. Other Personnel (list number in parentheses)		
1. (1) Post-doctoral associates	12 mos.	18,000
2. ( ) Other professional (technician, programmer, etc.)		
3. ( ) Graduate students		
4. ( ) Undergraduate students		
5. ( ) Secretarial - Clerical		
6. (1) Other		10,000
Total Salaries and Wages (A+B)		
C. Fringe Benefits (if charged as direct cost)		5,425
Total Salaries, Wages, and Fringe Benefits (A+B+C)		51,073
D. Permanent Equipment (list each item and dollar amount; insert lines if needed)		
1.		
2.		
3.		
Total Permanent Equipment		
E. Travel 1. Domestic		8,000
2. Foreign		
Total Travel		8,000
F. Trainee/Participant Costs (not applicable to NICCR)		**not applicable**
G. Other Direct Costs		
1. Materials and Supplies		8,000
2. Publication Costs		
3. Consultant Services		
4. Computer (ADPE) Services		
5. Subcontracts (total for all subcontracts)		
6. Other		12,164
Total Other Direct Costs		20,164
H. Total Direct Costs (A through H)		79,237
I. Total Indirect Costs (specify rates and bases) Total direct less fee remissions 51%		35,227
J. Total Direct and Indirect Costs (H+I)		114,464

DOE National Institute for Climatic Change Research: Budget Page Year Three

Organization: Indiana University/SPEA
Principal Investigator: Christopher Craft
Budget Start Date: 04/01/11 Budget End Date: 05/30/12

A. Senior Personnel	Person-months	Funds requested
1. Christopher Craft	1 mo/summer	9,436
2. Craig Wayson	2 mo/summer	8,742
3.		
4.		
5.		
6.		
B. Other Personnel (list number in parentheses)		
1. (1) Post-doctoral associates	12 mos.	18,000
2. ( ) Other professional (technician, programmer, etc.)		
3. ( ) Graduate students		
4. ( ) Undergraduate students		
5. ( ) Secretarial - Clerical		
6. (1) Other		10,000
Total Salaries and Wages (A+B)		
C. Fringe Benefits (if charged as direct cost)		5,620
Total Salaries, Wages, and Fringe Benefits (A+B+C)		51,798
D. Permanent Equipment (list each item and dollar amount; insert lines if needed)		
1.		
2.		
3.		
Total Permanent Equipment		
E. Travel 1. Domestic		3,000
2. Foreign		
Total Travel		3,000
F. Trainee/Participant Costs (not applicable to NICCR)		**not applicable**
G. Other Direct Costs		
1. Materials and Supplies		4,000
2. Publication Costs		2,000
3. Consultant Services		
4. Computer (ADPE) Services		
5. Subcontracts (total for all subcontracts)		
6. Other		11,672
Total Other Direct Costs		17,672
H. Total Direct Costs (A through H)		72,470
I. Total Indirect Costs (specify rates and bases) Total direct less fee remissions 51%		31,517
J. Total Direct and Indirect Costs (H+I)		103,987

BUDGET JUSTIFICATION

Salaries and Wages:

The PI and Project Director, Chris Craft, will spend 1 month/yr to work on the project. Co-PI, Craig Wayson (2 months per year), will be employed as a Research Associate and will focus his efforts on refining and running the SLAMM5 simulation model. A 3%/yr raise is included for the PI and Research Associate.

A full-time PhD student ($18,000/yr) will b employed to work on the project. Undergraduate student support ($4000-year 1, $10,000/yr –years 2 and 3) is requested for help with the field sampling and lab work.

Fringe Benefits:

PI:	21.50%
Research Associate:	21.50%
PhD student:	A 4%/yr increase in health insurance for the PhD student is included for years 2 and 3.

Permanent Equipment:

We request $25,103 to purchase a LICOR 8100 System ($16,800) for soil CO_2 flux measurements and Mac High-end Workstation ($8303) needed for SLAMM5 simulation modeling.

Travel:

We request $17,000 for travel to Georgia to collect data and present research results at national meetings. We will make seven trips to Georgia, including three trips in year 1 to select sites and collect data ($6,000) and four trips in year 2 to collect data from the field (soil transplant) and laboratory experiments ($8,000). $3,000 is requested in year 3 for one trip to Georgia and for the PI and PhD student to present research results at national scientific meeting ($1,000).

Each trip will cost approximately $2,000 for vehicle rental, lodging and per diem for one week or longer (for the field experiment) for four researchers

Materials and Supplies:

$19,000 is requested to cover materials and supplies for field collection of data, lab/analytical analyses and instrument time. In year 1, $6,000 is requested for field supplies ($500), lab supplies ($4000) and liquid nitrogen (LN) for gamma spectrometry ($1500).

In year 2, $8,000 is requested for field supplies for the soil transplant experiment ($1000), lab supplies ($4,000) for the lab experiment, LN ($1,000) and instrument time on the ratio mass spectrometer for $^{28}N_2$, $^{29}N_2$, and $^{30}N_2$ measurements of denitrification ($2,000).

We request $4,000 for supplies ($2,000), LN ($1,000) and instrument time on the RMS for measurement of $^{28}N_2$, $^{29}N_2$, and $^{30}N_2$ ($1,000) in year 3.

Boat Rental and Gas:

$6000 ($3000-year 1, $2000-year 2, $1000-year 3) is requested to cover boat rental and gas to collect samples from TFFF of the four rivers and to set up/collect data from the field (soil transplant) experiment.

Fee Remission:

Fee remission for in-state (IU) tuition is requested. A 5%/yr increase is included for years 2 and 3.

Publication Charges:

$2,000 (year 3) is requested to cover publication charges and reprints.

Indirect Costs: 51.5%

Abstract

Tidal freshwater floodplain forests (TFFF) exist at the nexus between terrestrial and marine influences and are among the most susceptible ecosystems to climate change, manifested as sea level rise and variation in freshwater input. Of coastal terrestrial ecosystems, TFFF are unique in that they provide water quality (WQ) improvement functions that may reduce nutrient loadings to and eutrophication of estuaries downstream. Using a combination of field measurements, manipulative experiments, geographic information systems (GIS) and simulation modeling, we will characterize the WQ improvement functions of TFFF and investigate how accelerated sea level rise (SLR) and varying freshwater discharge will alter their area and delivery of these functions. We hypothesize that:

I. Accelerated SLR during the next 100 years will reduce the area and, hence, delivery of WQ improvement functions from TFFF, through habitat loss and conversion as tidal marshes migrate inland.

II. Saltwater intrusion into TFFF will reduce denitrification and release inorganic N & P via desorption and increased anaerobic C mineralization, especially sulfate reduction.

III. Alteration of current freshwater (river) discharge regimes will influence the magnitude of impact from accelerated SLR: reductions in freshwater discharge will greatly reduce the area and delivery of WQ improvement function from TFFF, while increased freshwater discharge will offset some of the loss of water quality improvement functions caused by SLR.

IV. Thresholds exist within predicted ranges of SLR and freshwater discharge, after which TFFF area, ecosystem migration, and overall delivery of WQ improvement functions are markedly affected.

Water quality improvement functions (sediment deposition, N, P, organic C sequestration in soil, denitrification, inorganic N&P sorption/desorption) will be measured in TFFF of three rivers (Altamaha, Ogeechee, Satilla) of coastal Georgia (GA) and compared with measurements made in a degraded (i.e. currently experiencing saltwater intrusion) TFFF on the South Newport River. Soil cores from a TFFF will be transplanted into an oligohaline ($5^{o}/_{oo}$) marsh to test the effects of saltwater intrusion on denitrification, inorganic N&P desorption and soil respiration. A laboratory experiment will be used to test the effects of saltwater intrusion (0, 2 and $5^{o}/_{oo}$) and inundation (saturated, +10 cm) on denitrification, N&P desorption and C mineralization. The Sea Level Affects Marshes Model version 5 (SLAMM5) will be used to simulate changes in TFFF area and delivery of WQ improvement functions in response to inundation and saltwater intrusion caused by accelerated SLR and variable freshwater discharge during the next 100 years. Using National Wetland Inventory data, results will be scaled to the GA-SC coast.

Deliverables will include (1) characterizing WQ improvement functions of TFFF, ecosystems which are extremely susceptible to accelerated SLR and for which such data are lacking, (2) manipulative experiments to test the effects of saltwater intrusion and inundation on WQ improvement functions, (3) prediction of how SLR and variable freshwater discharge will interact to alter the area and delivery of WQ improvement functions at ecosystem and regional scales, and (4) advances to SLAMM5 that will allow for modeling the effects of both inundation AND saltwater intrusion in response to accelerated SLR and variable freshwater discharge that can be applied to other river-dominated coastal systems such as Chesapeake Bay, Hudson River and Gulf Coast rivers.

Introduction

Tidal freshwater floodplain forests (TFFF) are a unique feature of the coastal plain landscape of the southeast and Gulf coasts. They exist at the nexus between terrestrial freshwater discharge and tidal forcing from the sea and encompass approximately 72,000 ha in Georgia and South Carolina (C. Craft and J. Ehman, unpublished analysis of National Wetland Inventory data) with lesser amounts in North Carolina (>12,000 ha) (Hackney and Yelverton 1990) and Virginia (28,000 ha) (Rheinhardt and Hershner 1992). Because of their position at the uppermost reaches of river-dominated estuaries, they are extremely vulnerable to rising sea level and variation in freshwater river input driven by climate change (DeLaune et al. 1987, Pezeshki et al. 1990, Doyle et al. 2007).

Water Quality Improvement Services of Tidal Freshwater Floodplain Forests

Tidal freshwater floodplain forests and other tidal wetlands provide important ecosystem services to society. Such services include functions associated with regulation, habitat and production (de Groot et al. 2002) and specific examples include waste treatment/water quality improvement, floodwater storage, habitat for wild plants, biological productivity and recreation (Richardson 1994, Daily et al. 1997). Tidal freshwater floodplain forests support high levels of primary production (Fowler and Hershner 1989, Wharton 1978) and contain high plant species richness (Rheinhardt 1992) that supports a diverse animal community (Wharton et al. 1978). The importance of TFFF as sites of active nutrient cycling, retention and their role in water quality (WQ) improvement, however, is essentially unknown.

In a newly released book edited by Conner et al. (2007), a review paper by Anderson and Lockaby (2007) summarizes the limited data pertaining to soils and biogeochemistry of tidal freshwater forested wetlands of the southeastern U.S. but contains no information on WQ improvement functions of these wetlands. In the same book, Kroes et al. (2007) present some limited data regarding water quality functions of tidal and non-tidal reaches of the Pocomoke River, Maryland. In their study, sediment deposition on clay pads was similar in tidal (3.1 mm/yr) versus non-tidal (4.0 mm/yr) reaches. Deposition of organic matter was greater in the tidal reach whereas the non-tidal reach trapped mostly mineral matter. This study suggests that tidal and non-tidal floodplain forests store different types of sediment through different processes. However, the findings are based on a single river system and it underscores the paucity of data pertaining to WQ improvement functions of TFFF. There are, however, numerous studies pertaining to WQ improvement functions of other tidal wetlands (e.g. marshes) and non-tidal floodplain forests and, from these studies, we can make inferences regarding WQ improvement functions of TFFF.

It is well documented that tidal marshes and non-tidal floodplain forests intercept sediment and phosphorus (P), sequester nitrogen (N) in soil organic matter and denitrify nitrate in floodwaters primarily to N_2 (Conner and Day 1982, Brinson 1990, Craft and Casey 2000, Noe and Hupp 2005, Craft and Schubauer-Berigan 2006, Craft 2007). These wetlands are important for reducing N loads to estuaries and nearshore water which typically are N limited and, thus, susceptible to N enrichment and eutrophication (Howarth 1988, Howarth et al. 2002). Nitrogen sequestration in non-tidal floodplain soils of the southeast ranges from 47-78 kg N/ha/yr (Brinson et al. 1980, Craft and Casey 2000, Noe and Hupp 2005, Craft and Schubauer-Berigan 2006) with comparable amounts (2-46 kg N/ha/yr) removed by denitrification (Lowrance et al. 1984, 1985, Thompson et al. 1990). Floodplain forests receiving high nitrate (NO_3^-) loadings, either from anthropogenic sources or from coupling of nitrification with denitrification, remove as much as 90-430 kg N/ha/yr (Brinson et al. 1984, Poe et al. 2003). Tidal and floodplain wetlands are less effective sinks for P than for N. Non-tidal floodplain wetlands retain 2-32 kg P/ha/yr (Kuenzler et al. 1980, Brinson et al. 1980, Yarbro 1983, Craft and Casey 2000, Noe and Hupp 2005), mostly through sedimentation (Noe and Hupp 2005) and sorption to metal (aluminum-Al, iron-Fe) -organic matter complexes (Walbridge and Struthers 1993, Darke and Walbridge 2000).

Essentially nothing is known about the ability of TFFF to retain and remove N and P. These wetlands probably are important sinks for N because twice daily tidally-driven inundation promotes reducing (anaerobic) conditions that slows organic matter decomposition and promotes N sequestration in soil (Craft et al. 2002) and supports denitrification that removes N from the system (Hanson et al. 1994). Limited data from a TFFF (Clayhole Swamp) of the Altamaha River, Georgia, supports this hypothesis. Radiometric (^{137}Cs) dating of two soil cores reveals that the tidal freshwater floodplain forest sequesters comparable amounts of N and organic C as tidal freshwater marshes. The tidal freshwater floodplain forest sequesters somewhat less N and organic C relative to brackish marshes but much more than salt marshes in the estuary (Figure 1a, b). Phosphorus accumulation in soils of the tidal freshwater floodplain forest was much less relative to tidal freshwater and brackish marshes in the estuary (Figure 1c) and was attributed to reduced deposition of mineral sediment (Figure 1d) and particulate P.

Effects of Climate Change on Tidal Freshwater Floodplain Forests

Because they are located at the extreme upstream portion of the tide range, TFFF are vulnerable to accelerated rise in sea level that results in increased inundation and saltwater intrusion

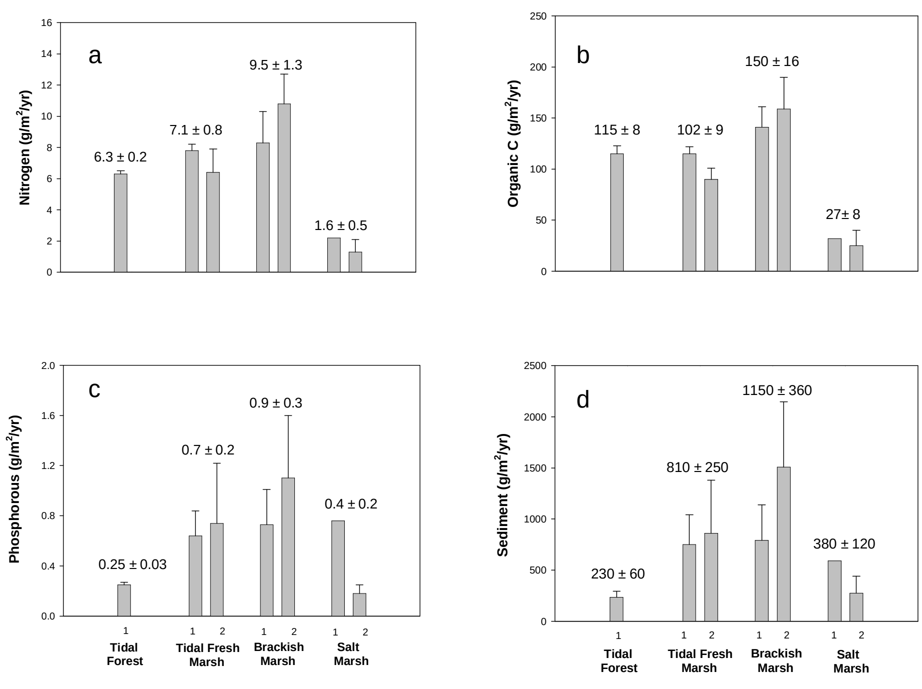


Figure 1. (a) Nitrogen, (b) organic carbon, (c) phosphorus and (d) mineral sediment accumulation in one tidal freshwater floodplain forest and two tidal fresh-, brackish- and salt-water marshes of the Altamaha River estuary, Georgia. Means (±1 SE) were calculated using ^{137}Cs-accretion, bulk density and total N, organic C, P concentrations in cores from levee and marsh plain locations (n=2 cores per marsh). Tidal marsh data are from Craft (2007) and TFFF data are from C. Craft, unpublished data.

(DeLaune et al. 1987, Pezeshki et al. 1990, Doyle et al. 2007). Global warming is projected to increase the rate of sea level rise (SLR), leading to habitat loss through submergence (Park et al. 1989, Brinson et al. 1995, Moorhead and Brinson 1995) and habitat conversion as ecosystems migrate landward (Park et al. 1991). And, recent evidence suggests that SLR has accelerated since 1993 with a globally-averaged rate of 3.3 ± 0.4 mm/yr (1993-2006) which is greater than the IPCC best-estimate rise of less than 2 mm/yr (Rahmstorf et al. 2007). Global warming also is projected to increase inter-annual variability of precipitation, leading to greater frequency of drought and floods (Karl et al. 1995, Mahlman 1997) and greater variability in freshwater discharge of rivers and streams. Reduced freshwater discharge may interact synergistically with rising sea level to increase saltwater intrusion and cause rapid loss of TFFF.

Saltwater intrusion is especially problematic for TFFF because, relative to tidal marshes, they depend more on soil organic matter than mineral sediment to maintain their elevation in the face of rising sea level (Figures 1b, d). As sea level rises, saltwater intrusion will increase sulfate concentrations that promote microbial sulfate reduction and enhance decomposition of soil organic matter (Weston et al. 2006, Craft 2007). Thus, TFFF are at greater risk for submergence than tidal marshes because, as sea level rises and saltwater intrudes, anaerobic decomposition (i.e. sulfate reduction) will consume soil organic matter that would normally support vertical accretion under freshwater conditions.

Saltwater intrusion also dramatically alters N and P cycles. In addition to mineralization of soil organic matter that releases NH_4^+ (Weston et al. 2006), nitrification is inhibited by sulfides (Joye and Hollibaugh 1995) and denitrification is reduced as sulfate reducers outcompete denitrifiers for available substrates (Weston et al. 2006). Also, increased salinity reduces sorption and increases efflux of inorganic N (NH_4^+) (Rysgaard et al. 1999), reducing soil exchangeable NH_4^+ and leading to reduced nitrification and denitrification in saline versus freshwater soils and sediments (Seitzinger et al. 1991, Rysgaard et al. 1999). Production of H_2S during sulfate reduction may directly inhibit denitrifiers, with the result that NO_3^- is converted to NH_4 via dissimilatory NO_3^- reduction rather than being converted to N_2O and N_2 during denitrification (Brunet and Garcia-Gil 1996, An and Gardner 2002). Saltwater intrusion also affects P cycling and retention by reducing sorption of P onto anion exchange sites (Sundareshwar and Morris 1999). And, HS^- produced during sulfate reduction binds with Fe to produce pyrite (FeS) and increases mobilization of sediment bound P (Caraco et al. 1989, Lamers et al. 2001).

These findings suggest that saltwater intrusion into TFFF will reduce N and P retention and removal and will, in fact, release these nutrients to the water column. This is problematic because loss and habitat conversion of TFFF and other freshwater wetlands will release N and P to downstream waters and exacerbate eutrophication of estuarine waters (Joye and Hollibaugh 1995).

Modeling the Effects of Sea level Rise on Tidal Freshwater Floodplain Forests

Along the southeast coast and elsewhere, SLR is projected to increase 30 to 100 cm in the next 100 years based on the International Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES) (Church et al. 2001, Meehl et al. 2007). As part of a US EPA-funded study in the region (www.spea.indiana.edu/wetlandsandclimatechange), the PI (Craft) leads a team investigating the effects of accelerated SLR on tidal marsh wetlands. Using a combination of field measurements, GIS, and simulation modeling using the Sea Level Affects Marshes Model, SLAMM (Park et al. 1986), the study investigates how area and delivery of eco

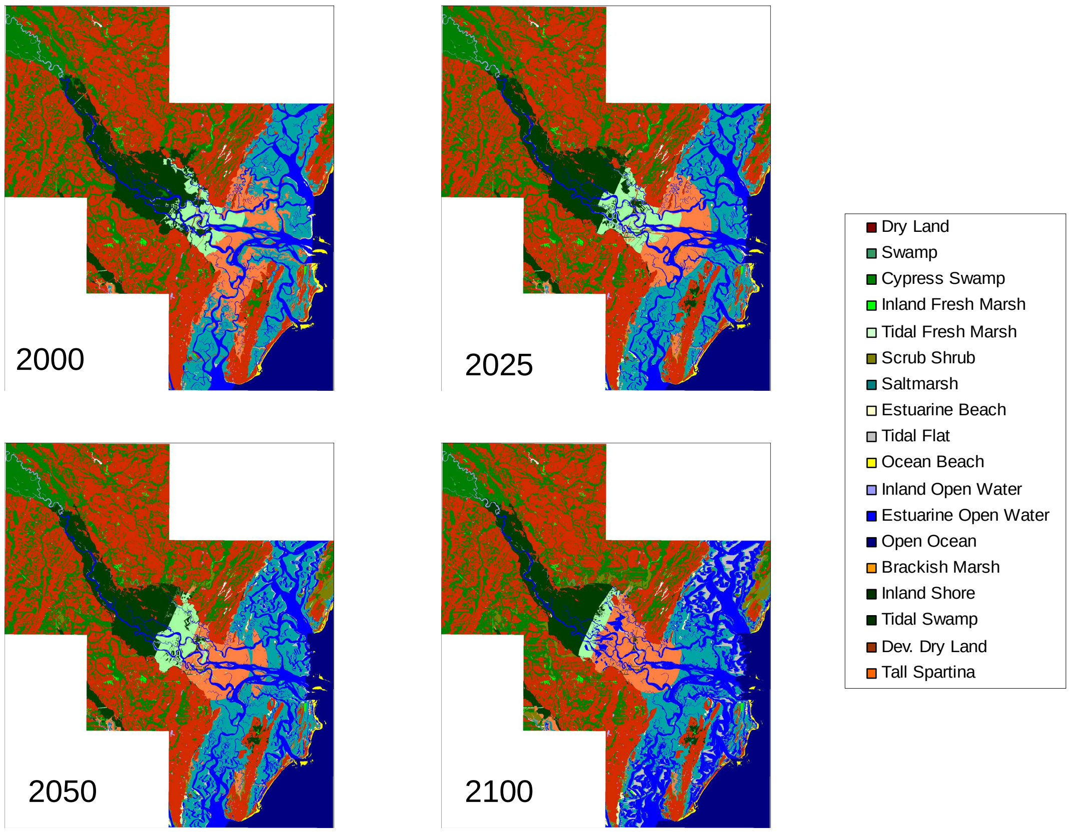


Figure 2. SLAMM5 simulation of the effects of sea level rise on land cover types in the Altamaha River Estuary (GA) in response to the A1B SRES sea level rise scenario that predicts a 70 cm increase in sea level by 2100, with no deviation from the current freshwater discharge regime.

system services of salt, brackish and tidal freshwater marshes will be altered by SLR (Figure 2). SLAMM was developed as an elevation-based model to predict how rising sea level will inundate coastal shorelines, wetlands and terrestrial ecosystems. As part of the EPA project, senior modeler and SLAMM developer Dick Park and team programmer Jonathan Clough extended SLAMM, now in version 5 so as to incorporate a salinity algorithm that will simulate saltwater intrusion, in addition to inundation, as sea level rises in river-dominated estuaries such as the Altamaha River and other estuaries where TFFF are mostly found.

Based on SLAMM5, preliminary results from the Altamaha River estuary, the largest river in Georgia and the third largest river on the east coast suggest that, with no alteration in freshwater river discharge, TFFF in the watershed will decline 20% during the next 100 years (Table 1). The model also predicts a loss of 10% of terrestrial land and a small increase (+2%) in non-tidal floodplain forest in the watershed as sea level rises (Table 1). Overall, the model predicts a net loss of non-tidal fresh marsh (-12%) and tidal marsh habitat, especially tidal freshwater marsh, with a corresponding increase in tidal flat and open water as they migrate inland (Table 1). SLAMM simulations of the Ogeechee and Satilla River estuaries (Georgia) show

Table 1. Predicted change in area (in hectares) of selected in (wet)land cover types of the Altamaha River in response to the A1B SRES sea level rise scenario as modeled by SLAMM5, with freshwater discharge held constant. Initial condition = 2000 AD

	Initial. Condition	Year 2100	Percent Change
Dry land	69,400	60,300	-13
Freshwater marsh	900	800	-12
Non-tidal floodplain forest	30,400	31,000	+2
Tidal fresh floodplain forest	**14,100**	**11,300**	**-20**
Tidal freshwater marsh	4,000	1,600	-61
Brackish marsh	10,000	8000	-20
Salt marsh	21,000	16,300	-22
Tidal flat	60	3300	+5500
Estuarine open water	11,300	24,500	+215

comparable declines in TFFF area, 9% and 32% respectively, in response to the IPCC SRES A1B sea level rise scenario (C. Craft and J. Ehman, unpublished data).

Net loss of TFFF and other tidal wetlands in the area is predicted to lead to a corresponding reduction in WQ improvement functions. In the SLAMM simulation of the Altamaha River estuary (Figure 2), based on direct measurements (Figure 1), we estimate that N accumulation in soil will be reduced by more than 600 MT yr^{-1}, with nearly 30% of the reduced retention attributed to loss of TFFF habitat (Table 2) that, based on the SLAMM5 simulation, will decline in area by 20% (Table 1). Carbon sequestration is reduced nearly 10,000 MT yr^{-1}, again with TFFF accounting for about 30% of the overall reduction (Table 2). Because of low P accumulation in TFFF relative to tidal marshes, loss of TFFF does not reduce P retention to the same degree as for N. A 20% reduction in TFFF area accounts for only about 10% of the overall reduction (7 MT yr^{-1}) in P retention in the estuary (Table 2).

Effects of Variable Freshwater Discharge on Tidal Freshwater Floodplain Forests

General circulation models predict warmer global temperatures and greater inter-annual variability in precipitation in response to increased CO_2 emissions (Cusbasch et al. 2001). And, regional model simulations for the southeastern U.S. suggest that temperature will increase 3°C to 5°C in response to 2 times the current levels of atmospheric CO_2 with the greatest warming in southeastern coastal states, GA, SC, NC and VA (Mearns et al. 2003). Precipitation patterns also are predicted to change with increased precipitation in spring but reduced precipitation, by up to 30%, in summer (Mearns et al. 2003). The combination of increased temperature and reduced rainfall during the summer is likely to lead to significant reduction in freshwater discharge of rivers along the southeastern coast during the growing season.

With warmer temperatures, greater evapotranspiration and reduced rainfall, freshwater discharge from rivers of the southeast coast could be reduced 25% or more. Reduced freshwater discharge combined with accelerated SLR could hasten the rate of habitat conversion as TFFF is replaced by more saline tidal “fresh” and brackish marshes. Moreover, habitat conversion and ecosystem migration may not occur gradually. Rather, the interaction between tidal inunda

Table 2. Predicted change in water quality improvement functions of tidal freshwater floodplain forests and tidal marshes of the Altamaha River estuary between 2000 and 2100 based on SLAMM5 simulations of accelerated SLR with freshwater discharge held constant. Accumulation rates were measured directly and are shown in figure 1.

Wetkland Type	Change in Area (ha)	Change in Water Quality Improvement Function (MT/yr)			
		Nitrogen	Phosphorus	Carbon	Sediment
Tidal fresh forest	**-2800**	**-175**	**-7**	**-3,200**	**- 6,400**
Tidal fresh marsh	-2400	-170	-17	-2,450	-19,400
Brackish marsh	-2000	-190	-18	-3,000	-23,000
Salt marsh	-4700	- 75	-19	-1,300	-17,900
Net change	-11900	-610	-71	-9,950	-66,700

tion, saltwater intrusion and reduced freshwater discharge may lead to thresholds of change, where habitat conversion and ecosystem migration occur abruptly.

Hypotheses:

Climate change is expected to alter the two primary drivers of TFFF hydrology- sea level rise and freshwater river discharge (Figure 3). Sea level rise will lead to reduced tidal freshwater floodplain forest habitat and WQ improvement functions through the combined effects of increased inundation and salinity (Figure 3). Reduced freshwater river discharge will act synergistically with SLR to greatly decrease the area and WQ improvement functions of TFFF (Figure 3). In contrast, if freshwater discharge increases, then the SLR-induced losses of TFFF habitat and the associated WQ improvement functions will be somewhat offset by reductions in saltwater intrusion.

I. Accelerated SLR during the next 100 years will reduce the WQ improvement functions provided by TFFF as tidal wetlands, forests and marshes, are replaced by open water and tidal flats.

II. Saltwater intrusion into TFFF soils will reduce denitrification and release NH_4-N and PO_4-P. Emissions of CO_2 will increase and CH_4 will decline in response to increased supply of marine sulfate.

III. Alteration of current freshwater (river) discharge regimes will influence the magnitude of impact from accelerated SLR: reductions in freshwater discharge will greatly reduce the area and delivery of WQ improvement functions from TFFF, whereas increased freshwater discharge will offset some of the loss of WQ improvement functions caused by SLR.

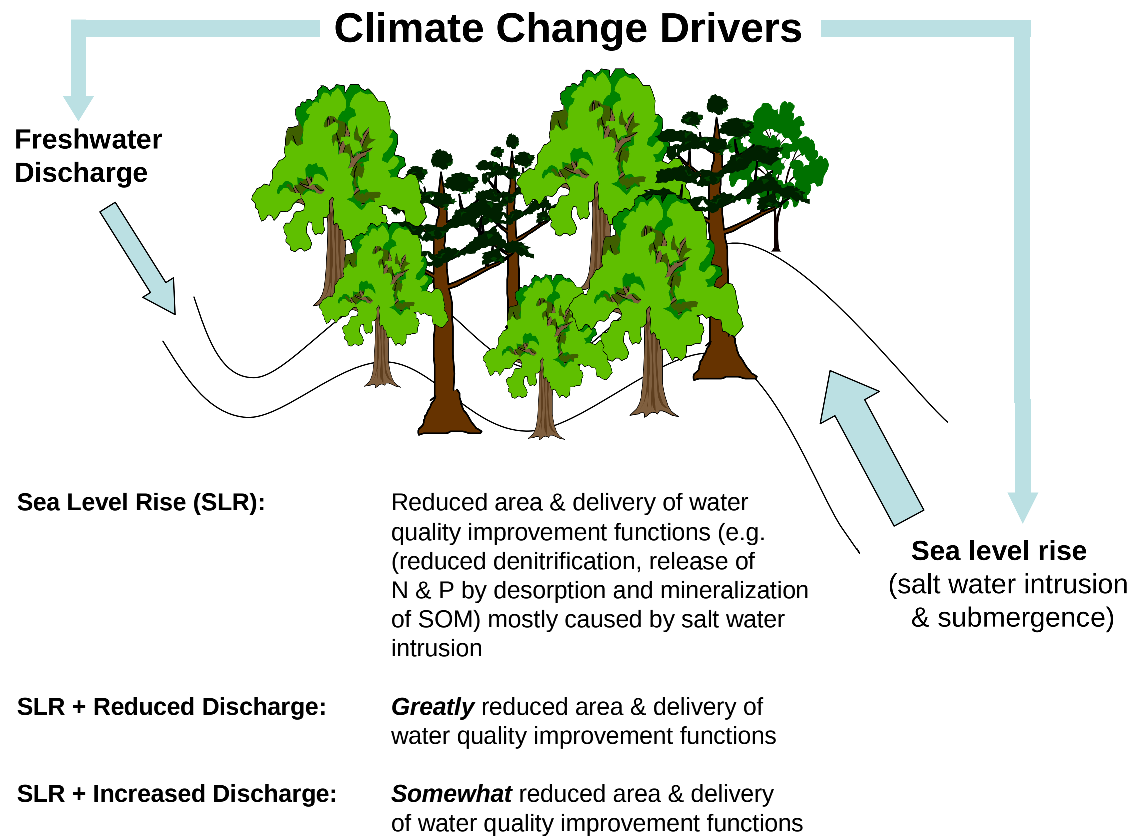


Figure 3. Hypothesized effects of climate change (sea level rise, freshwater discharge) on delivery of water quality improvement functions of tidal freshwater floodplain forests.

IV. Thresholds exist within predicted ranges of SLR and freshwater discharge, after which TFFF, ecosystem migration, and delivery of WQ improvement functions are markedly affected.

It has been suggested that coastal wetlands and terrestrial ecosystems exist at multiple stable states (Brinson 1995) such that, as sea level rises, abrupt transitions occur as one stable state shifts to another state. Increasing inundation and salinity drive these shifts as, for example, TFFF is replaced by tidal freshwater marsh, and tidal freshwater marsh is replaced by brackish and saltwater marsh (Moorhead and Brinson 1995, Brinson 1995). We will test this hypothesis by modeling incremental increases in sea level rise and variable (increased, decreased) freshwater discharge to identify thresholds associated with increased inundation, saltwater intrusion and freshwater river flow.

Methods

Site Description

Water quality improvement functions will be measured in TFFF of the Altamaha, Ogeechee and Satilla Rivers, Georgia (Figure 4). The Altamaha River is the third largest river on the east coast and is the largest river in Georgia with a drainage area of over 36,000 km^2 and annual discharge of 400 m^3/sec (Table 3). The Ogeechee and Satilla Rivers, after the Altamaha River, are the largest rivers whose watersheds are located entirely within the state of Georgia. The three rivers contain 92% (28,800 ha) of the TFFF in Georgia, and 22% of TFFF area in the southeastern U.S., South Carolina (49,000 ha), North Carolina (12,500 ha) and Virginia (28,000 ha) and Georgia combined (C. Craft unpublished data for GA and SC, Hackney and Yelverton 1990 for NC, Rheinhardt and Hershner 1992 for VA).

Two study sites each will be established in TFFF on each river. We will focus our field sampling efforts on 60-70 year-old forests dominated by bald cypress and tupelo gum. Flood-plain forests of the southeast coast were logged in the 1930's and 1940's so, today, this is the most prevalent age class of forest found in tidal reaches of the three rivers (C. Craft, personal observation). Two study sites also will be established on the South Newport River (Figure 4). These sites contain tidal floodplain forest that currently is experiencing saltwater intrusion, causing tree morality and habitat conversion to brackish marsh (C. Craft, personal observation). Measurements made at these sites will be used to gauge the effects of saltwater intrusion on WQ improvement functions in the field under "ambient" environmental conditions.

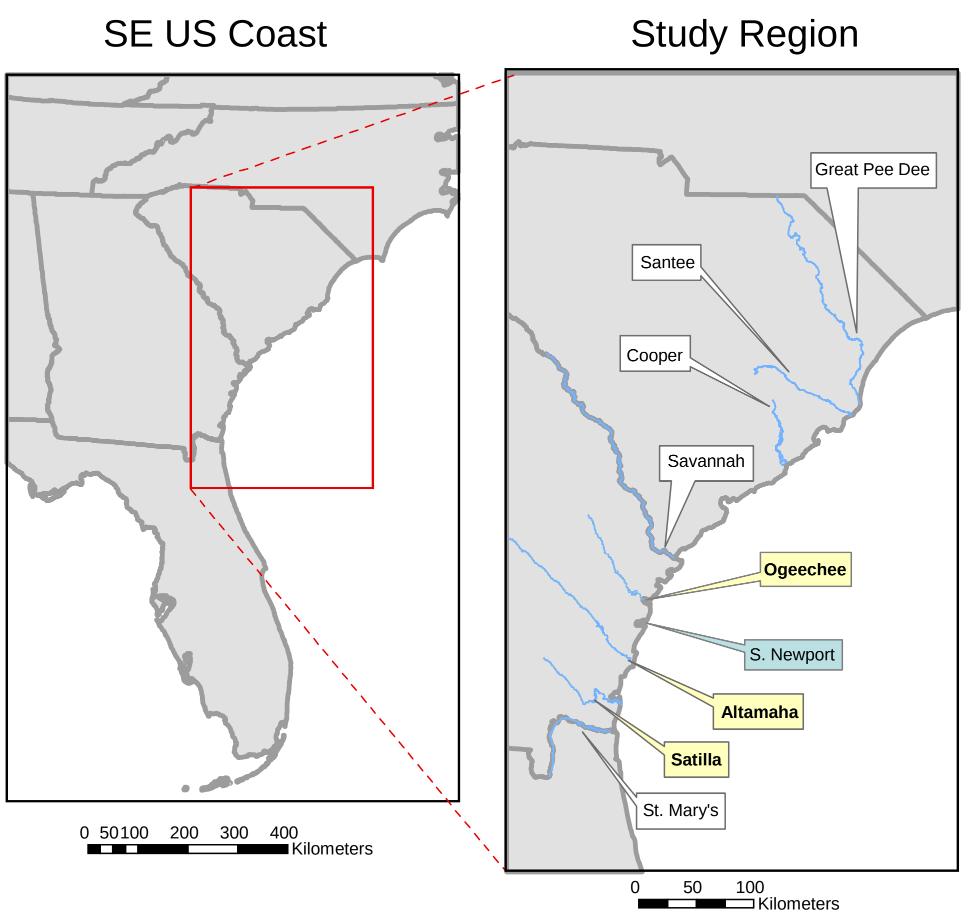


Figure 4. Location of field study sites (Altamaha, Ogeechee, Satilla Rivers) and other river systems that contain extensive tidal freshwater floodplain forests. The South Newport River contains tidal floodplain forest that currently is experiencing saltwater intrusion.

Table 3. Watershed size, river discharge, and area of tidal freshwater floodplain forest of the study sites. Watershed and discharge data are from http://lmer.marsci.uga.edu. Area of tidal freshwater floodplain forest is from an analysis of US FWS National Wetlands Inventory data (C. Craft and J. Ehman, unpublished data).

River/State	Watershed Size (km^2)	Discharge (m^3/sec)	Tidal fresh floodplain Forest (ha)
Altamaha	37,600	400	14,100
Ogeechee	7,000	100	5,500
Satilla	9,143	85	6,750

Water Quality Improvement Functions of Intact & Degraded Tidal Freshwater Floodplain Forest

Water quality improvement functions will be determined by measurements of sediment deposition, N&P accumulation in soil, denitrification and N&P sorption/desorption. Sediment deposition and N & P accumulation in soil will be measured by collecting two soil cores (8.5 cm diameter by 50 cm deep) from each site (n=16 cores), one each on the levee and in the floodplain interior. Cores will be sectioned into 2 cm increments and each increment will be analyzed for ^{137}Cs, ^{210}Pb, bulk density, N and P (Craft and Casey 2000) to determine forty (^{137}Cs) and 100 (^{210}Pb) year rates of sediment deposition and nutrient accumulation. Three 0.25 m^2 feldspar marker layers at each site in year 1 and small diameter soil cores (n=2 per plot) will be collected six and 12 months later to determine short-term rates of sediment accretion and accumulation (Cahoon 1994).

Denitrification will be measured by the isotope pairing technique, using ^{15}N enriched nitrate as part of the background ion matrix (Steingruber et al., 2001). Replicate cores (n=5 on the levee and 5 in the interior, 5 cm diameter by 20 cm deep) will be collected from each site and incubated using the batch-mode assay method (4 rivers x 2 forests x 10 samples = 80 assays). Evolved N_2 will be measured for stable isotopes, ($^{28}N_2$, $^{29}N_2$ and $^{30}N_2$), using continuous-flow isotope ratio mass spectrometry at the Stable Isotope/ Biogeochemistry Lab in the Department of Geological Sciences at Indiana University. This method has the advantage of being able to differentiate between denitrification of water column NO_3^- and coupled nitrification-denitrification (Herrman and White, in review; Steingruber et al., 2001).

N&P sorption/desorption will be determined by collection of ten (5 levee, 5 interior), 5 cm ID by 10 cm deep, intact soil cores from each site using PVC pipe. Rubber stoppers will be placed in each end of the pipe and cores will be transported to the lab on ice. In the lab, cores will be flooded to a depth of 5 cm with river water containing 30 μM NH_4-N and 5 μM PO_4-P to approximate the 6:1 NH_4:HPO_4^{2-} ratio found in Altamaha River water (Weston et al. 2006). Cores will be allowed to equilibrate for the equivalent of one flooding cycle (6 hours), after which the surface water will be removed and an aliquot analyzed for NH_4-N and PO_4-P(APHA 1998). The same protocol will be performed four more times to characterize inorganic N and P sorption/desorption over five simulated tidal cycles (4 rivers x 2 forests x 10 samples x 5 tidal cycles = 400 assays).

Statistical Analyses

A factorial analysis of variance (ANOVA) based on river system (Altamaha, Savannah and Satilla Rivers; S. Newport River), and sampling location (levee, interior) will be used to compare WQ improvement functions (sediment deposition, N and P in soil, denitrification, N&P sorption/desorption) of intact TFFF and to test for differences between intact forests and the S. Newport River forest that is experiencing saltwater intrusion (SAS 1996). Repeated measures ANOVA will be employed to test for differences in N&P sorption/desorption among sites and within site locations. Where appropriate, data will be transformed to meet the assumptions of the ANOVA (Sokal and Rohlf 1995). Typically, proportional data will be arcsine (square root) transformed, and in cases where the variance increases with the mean, numerical data will be log transformed, in order to improve normality and homogeneity of variance. Means will be separated using a posteriori means comparison tests such as the Ryan-Einot-Gabriel-Welsch (REGW) multiple range test (SAS 1996).

<u>Effect of Saltwater Intrusion on Water Quality Improvement Functions</u>

Field Experiment

A field experiment whereby intact soil cores from a TFFF are transplanted to an oligohaline ($5^{o}/_{oo}$) marsh will be used to test the hypotheses that saltwater intrusion reduces denitrification and releases soil inorganic N and P. One hundred intact soil cores (8.5 cm diameter by 30 cm deep) will be collected from a TFFF of the Altamaha River. Fifty cores will be transplanted to the oligohaline ($5\ ^{o}/_{oo}$) marsh downstream and the remaining 50 cores will be placed back into their respective holes to test for the potential effects of disturbance. At intervals of 1 week, one month, 2 months and 4 months following transplantation, 10 transplanted cores at each site will be “sacrificed” by removing two 3 cm diameter by 10 cm deep soil cores, one for denitrification and one for N&P sorption/desorption (n=10 for each sampling date). The small diameter cores will be returned to the lab and analyzed for denitrification and N&P sorption/desorption as described previously (2 treatments x 10 samples x 4 sampling dates = 80 assays).

The 10 cores remaining at each site will be used for non-destructive measurements of soil respiration. During each sampling event, in-situ soil respiration will be measured using a LICOR model 8100 infrared gas (CO_2) analyzer and soil respiration chamber. Ten additional respiration measurements per sampling event will be made in undisturbed soils of the tidal freshwater floodplain forest to gauge the potential effect of soil disturbance caused by transplanting the soil cores (3 treatments x 10 samples x 4 sampling dates = 120 measurements).

A one-way ANOVA will be used to test the effects of saltwater intrusion on denitrification, N&P sorption/desorption, and the potential effects of soil disturbance on these processes (SAS 1996). Repeated measures ANOVA will be employed to test for the effects of saltwater intrusion and disturbance on soil respiration. Where appropriate, data will be transformed to meet the assumptions of the ANOVA (Sokal and Rohlf 1995). And, means will be separated using a posteriori means comparison tests such as the Ryan-Einot-Gabriel-Welsch (REGWQ) multiple range test (SAS 1996).

Laboratory Experiment

The field-based soil transplant experiment will be supplemented by a lab-based soil incubation experiment to further investigate the effects of saltwater intrusion as well as inundation on denitrification, N&P sorption/desorption and anaerobic C mineralization. We will conduct soil incubation experiments in a fully crossed design using soil cores from one TFFF site on each river system. Treatments will consist of salinity levels of 0, 2 and $5^{o}/_{oo}$, with a salt matrix that matches sea water ion ratios and inundation levels of 0 cm and +10 cm. Anaerobic conditions will be maintained and temperature and pH controlled during incubations. Rates of denitrification will be measured using the isotope pairing technique, using ^{15}N enriched nitrate as part of the ion matrix for each treatment as described previously (Steingruber et al., 2001).

Rates of CO_2 and CH_4 production will be measured over a period of several months to quantify changes in organic carbon mineralization (see Keller et al., 2004). Enhanced rates of bacterial sulfate reduction are likely to be a key control on organic carbon mineralization rates so a passive method whereby total reduced inorganic S is extracted will be used to quantify rates of sulfate reduction for each core (Ulrich et al. 1997).

A total of 54 cores per river [3 salinity x 2 inundation x (3 denitrification + 3 C mineralization + 3 sulfate reduction)] will be used in the Lab Experiment for a total of 162 cores.

Statistical analysis of data will be performed using repeated measures ANOVA based on salinity and inundation regimes (SAS 1996). Where appropriate, data will be transformed to meet the assumptions of the ANOVA (Sokal and Rohlf 1995). And, means will be separated using a posteriori means comparison tests such as the Ryan-Einot-Gabriel-Welsch (REGWQ) multiple range test (SAS 1996).

Modeling the Interactive Effects of Accelerated SLR and Freshwater River Discharge

Changes in TFFF area and delivery of WQ improvement functions in response to different scenarios of sea level rise will be modeled using the Sea Level Affects Marshes Model (SLAMM) developed by Dick Park for USEPA (Park et al. 1986). Our estimates of sea level rise are taken from climate change models that are reported in the IPCC Special Report on Emissions Scenarios (SRES) (Church et al. 2001). Our predicted estimates of sea level rise range from 30 cm to 100 cm (mean=70 cm) by the year 2100 and are based on SRES A1 which assumes rapid economic growth, low population growth and rapid introduction of new and more efficient technology. Our modeling scenarios are consistent with the most recent (2007) IPCC findings based on the A1B SRES, in which global sea level rise is predicted to increase to 3.8 mm/yr between 2090 and 2100 (Meehl et al. 2007). The SRES A2 scenario which assumes a lower rate of economic growth, fewer technological advances but greater population growth, predicts a similar increase in sea level rise during the next century (Church et al. 2001).

SLAMM simulates the dominant processes involved in wetland conversions and shoreline modifications during long-term sea level rise. A complex decision tree incorporating geometric and qualitative relationships is used to represent transfers among coastal classes. Each site is divided into cells of equal area, and each class within a cell is simulated separately. SLAMM uses spatially-explicit data inputs that include the USFWS National Wetland Inventory, NOAA tidal data, USGS streamflow data, and the USGS National Elevation Data (NED). The latter dictates the raster-based model's typical cell resolution: ~30 m. All of these inputs have been acquired and vertically integrated as part of the EPA-funded study of tidal marshes. Distributions of wetlands and uplands are predicted under conditions of accelerated sea level rise, and results are summarized in tabular and geospatially-referenced "map" form.

Relative sea level change is computed for each site for each time step; it is the sum of the historic eustatic trend, the site-specific rate of change of elevation due to subsidence and isostatic adjustment, and the accelerated rise depending on the scenario chosen (Titus et al. 1991). Sea level rise is offset by sedimentation and accretion using site-specific radiometric (^{137}Cs, ^{210}Pb)

measurements to be collected in conjunction with N and P accumulation in soil. For each time step the fractional conversion from one class to another is computed on the basis of the relative change in elevation divided by the elevational range of the class in that cell.

As part of the EPA-funded study that investigates the effects of SLR on ecosystem services provided by tidal marshes, we modified SLAMM (Version 4) to simulate the effects of saltwater intrusion on habitat conversion and ecosystem migration of wetlands Similar to previous versions, SLAMM5 models inundation using elevation data but differs in that it uses a salinity algorithm to drive the saltwater wedge upstream as sea level rises in river-dominated estuaries suchas the Altamaha River (see Figure 2). Designed to incorporate both rising sea level and freshwater river discharge of varying magnitudes, the SLAMM5 salinity routine is especially useful for modeling SLR in river-dominated estuaries and coastal river systems of the southeast. There, surface elevation of tidal wetlands (brackish and fresh marsh, floodplain forest) varies little, (+1.3 to +1.6 m MSL based on NED and limited LIDAR data; C. Craft, unpublished data) and saltwater intrusion rather than submergence leads to habitat conversion and ecosystem migration. Outside of river-dominated coastal systems where freshwater input is minimal and TFFF are absent, SLAMM5 uses elevation data (described above for earlier versions of SLAMM) to simulate the effects of sea level rise.

SLAMM5 will be employed to model incremental (10 cm) changes in sea level rise up to 1 m as well as extreme increases in sea level (1.5 m, 2 m) to identify thresholds of abrupt increases in submergence, habitat conversion, ecosystem migration and changes in delivery of WQ improvement functions of TFFF. As part of the DOE proposal, the interactive effects of freshwater discharge and sea level rise on area and delivery of WQ improvement functions of TFFF will be simulated. Since SLAMM5 incorporates river discharge as part of the salinity algorithm, varying freshwater discharge will modify the rate at which saltwater intrudes, TFFF habitats undergo conversion and ecosystems migrate. Regional GCM's predict that, in the southeastern U.S. temperatures will increase 3-5^{o}C and summer precipitation will decline by 30% with 2X CO_2 (Mearns et al. 2003) so that freshwater discharge may decrease by 25-50% during the growing season. We will model variable freshwater discharge in increments of 10%, from -50% to +50% of the long-term mean (1930's to present) (Hickey et al. 2001) to simulate the effects of both increased and decreased freshwater discharge.

Predicted changes in TFFF area will be combined with measurements of WQ improvement functions to quantify changes in their delivery under different scenarios of sea level rise and freshwater discharge. Measurement of a comparable suite of WQ improvement functions (i.e. sediment deposition, N, P, organic C accumulation in soil, denitrification) of salt, brackish and tidal freshwater marshes, funded by EPA, will enable us to evaluate the overall net change in delivery of wetland WQ improvement functions for entire the tidal riverine/estuarine wetland complex as sea level rises and estuarine marshes migrate into areas currently occupied by tidal freshwater and non-tidal floodplain forests.

Model Validation

We will use two published studies, both within our study domain, to validate the results of our SLAMM simulations. On the Altamaha and Satilla Rivers, changes in tidal wetland vegetation between 1953 and 1993 documented by Higginbotham et al. (2004) in conjunction with the historical increase in sea level rise for the period will be used determine whether SLAMM5 correctly predicts spatial changes in tidal wetland plant communities for the period. Simulation results from the two historical studies will enable us to determine to what degree SLAMM5 accurately forecast the effects of future sea level rise on changes in tidal wetland area, habitat conversion and ecosystem migration during the next 100 years.

In another study on the Savannah River, a tide gate was constructed in 1977 to scour part of the river channel and reduce maintenance dredging (Pearlstine et al. 1993). An unintended consequence was a nearly six mile displacement of the salt wedge upstream, leading to increased salinity and change in tidal fresh species to species associated with brackish marsh habitat (Pearlstine et al. 1993). Using GIS coverage from the EPA study, we will use SLAMM5 to reproduce the corresponding increase in salinity in the river to determine how well the model simulates the change from tidal freshwater vegetation to brackish marsh vegetation that was documented by Pearlstine et al. (1993).

<u>Scaling from Watershed to Southeastern (GA-SC) Coast</u>

Measurements of WQ improvement functions at the plot level will be scaled to the extent of the study region (GA-SC) using the NWI-based SLAMM5 inputs previously acquired as part of the EPA-funded study, and the incremental SLAMM5 outputs, using a per unit area approach. The areas of TFFF class are known for the SLAMM5 model (baseline) inputs and will be calculated for the SLAMM5 outputs associated with each incremental rise in sea level and variation in freshwater discharge. This information enables the various WQ improvement functions (i.e. sediment deposition, N, P, organic C accumulation in soil, denitrification, N&P sorption/ desorption) to be directly scaled to the extent of the study region, such that changes in delivery of WQ improvement functions associated with different incremental increases in sea level and variation in freshwater discharge can be calculated. Moreover, the incremental modeling approach will allow us to identify thresholds of change in TFFF area and delivery of WQ improvement functions associated with specific scenarios of SLR and freshwater discharge.

Relevance to DOE NICCR Research Goals

The proposed work will quantify WQ improvement functions (sediment deposition, N and P accumulation in soil, denitrification, N&P sorption/desorption) of TFFF. Of coastal terrestrial ecosystems, TFFF are unique in that they supply a key ecosystem service (i.e. water quality improvement) that has the potential to reduce pollutant loadings to downstream estuaries, tidal marshes and nearshore waters. Nitrogen retention and removal, in particular, is of critical importance since many estuaries and tidal wetlands are N limited and, thus, are susceptible to N eutrophication (Howarth 1988, Howarth et al. 2002, Frost et al., in review).

Tidal freshwater floodplain forests are extremely vulnerable to the effects of climate change, sea level rise and variable river discharge (DeLaune et al. 1987, Pezeshki et al. 1990, Doyle et al. 2007) yet, to date, no systematic effort has been undertaken to identify the kinds and magnitude of WQ improvement functions provided by these wetlands. Accelerated SLR may exacerbate N loadings to estuaries by (1) inundating TFFF and other tidal wetlands that lead to a reduction in the N and P assimilative capacity of the watershed (see, for example, Table 2) and (2) saltwater intrusion and sulfate reduction that releases sorbed inorganic N and P (Seitzinger et al. 1991, Sundareshwar and Morris 1999), inhibits nitrification-denitrification (Joye and Hollibaugh 1995, Rysgaard et al. 1999) and accelerates decomposition of soil organic matter (Weston et al. 2006).

Deliverables from this study will consist of (1) quantifying WQ improvement functions provided by TFFF, (2) assessing the effects of saltwater intrusion on WQ improvement functions determined by both field measurements and (soil) transplant experiments, (3) differentiating between the effects of salinity versus inundation on a subset of WQ improvement functions in a lab study, (4) simulation modeling to identify change in the area, delivery of WQ improvement functions, and thresholds of change of TFFF of the three river systems and the south Atlantic coast (GA-SC) in response to rising sea level, including the effects of both inundation and saltwater intrusion, and (5) refinements to SLAMM5 to simulate the interactive effects of sea level rise and variation (increased, decreased) in freshwater discharge on TFFF habitat and ecosystem migration that can be applied to river-dominated coastal floodplain systems elsewhere.

In addition, the investigators are committed to providing for public *discovery* of the project and *accessibility* and *evaluation* of results. A project web-site, to be developed and hosted at Indiana University, will include a project description, workplan, results as they become available, links to the researchers, and acknowledgement of the DOE National Institute for Climatic Change Research including display of appropriate logo(s). The web site will feature a simple interactive web-mapping application that presents the geospatial project results (i.e., SLAMM5 output and scaled ecosystem service layers) displayed with contextual information (e.g., administrative boundaries, roads, and geographic names). FGDC-compliant metadata will be created for all geospatial products, and the project and site will be registered via DIF-metadata in the Global Change Master Directory (GCMD). A project archive will be made available upon its conclusion.

General Project Information

Project Organization

Chris Craft (Indiana University) will serve as PI and will oversee site selection, coordination of field sampling, field and lab experiments, and SLAMM5 modeling. He also will supervise the soil/sediment accumulation, soil transplant experiment including in situ soil respiration measurements, and N&P sorption/desorption work. The Craft Lab is equipped with gamma spectrometer for ^{137}Cs and ^{210}Pb analysis, Perkin-Elmer CHN analyzer, UV/visible & atomic absorption spectrophotometers and wet lab space for soil digestion. Co-PI Jeff White, who is on a 12-month academic appointment, cannot accept salary but will contribute to the project by supervising the denitrification measurements and the laboratory salinity/inundation experiment. His Lab is equipped with gas chromatographs for CO_2 and CH_4 measurements. Both the PI (Craft) and Co-PI (White) are members of the Geology Biogeochemistry Laboratories at Indiana University and have access to the Stable Isotope Lab where the denitrification ($^{28}N_2$, $^{29}N_2$ and $^{30}N_2$) analyses will be conducted.

Craig Wayson (Co-PI) will supervise SLAMM5 simulations of accelerated SLR and variation (increased, decreased) freshwater river discharge of the three river systems and the south Atlantic (GA-SC) coast. Craig also will maintain the GIS database and website. If funded, Indiana University (SPEA) has agreed to provide $10,000 funding to support Jonathan Clough, SLAMM programmer, to provide technical support as needed, for troubleshooting and fine-scale refinements to the model. A letter of support from Dick Park, who developed SLAMM, is contained in appendix A.

Synergies with Other Research

The proposed work integrates strongly with ongoing climate change studies in the region. The most robust link is with the USEPA Science to Achieve Results (STAR) grant *Effects of Sea Level Rise and Climate Variability on Ecosystem Services of Tidal Marshes* awarded to the PI

(Craft), SLAMM developer (Dick Park) and SLAMM programmer (Jonathan Clough) (www.spea.indiana.edu/wetlandsandclimatechange). The project consists of field-based measurements of ecosystem services provided by salt, brackish and tidal freshwater marshes and the effects of accelerated SLR, using SLAMM, on marsh area, ecosystem migration and delivery of ecosystem services. The DOE proposal, if funded, will use existing NWI and elevation data and SLAMM refinements, including the salinity algorithm, developed from the EPA STAR grant to extend our simulation modeling of tidal marshes upriver into TFFF and non-tidal floodplain forests. Additionally, the EPA supported research adds value to our DOE proposal because it enables us to predict the effects of SLR and altered freshwater discharge on the *net* delivery of tidal wetland WQ improvement functions in the region as tidal marshes, and their WQ improvement functions, migrate inland and replace TFFF.

Another synergy that can be used to leverage resources for the proposed DOE work is the Georgia Coastal Ecosystems Long Term Ecological Research (GCE LTER) project funded by the National Science Foundation. The GCE LTER project, of which Chris Craft is Co-PI, was initiated in 2000 and was renewed for a second 6-year funding cycle in November 2006 (http://gce-lter.marsci.uga.edu/lter). The overarching goal of the GCE LTER is to understand how freshwater input, precipitation and river discharge, structures tidal marsh, estuarine and near-shore processes. The GCE LTER can provide logistical support through the University of Georgia Marine Institute, based at Sapelo Island, Georgia, and by collection and access to long-term monitoring data of climate, populations of biota, and ecosystem processes of the region.

As part of our regional modeling efforts, the PI (Craft) also works with Brian Czech of the US Fish and Wildlife Service (USFWS) to forecast the potential impacts of sea level rise on USFWS Reserves in the southeast (GA-SC) region. Our modeling results will be used by regional managers to develop Reserve-specific strategic plans to respond to sea level rise and saltwater intrusion during the next 20-100 years.

Finally, the PI (Craft) works closely with The Nature Conservancy (TNC) on several projects in the Altamaha River watershed, which is designated as a biodiversity "hotspot" by TNC. Chris Craft currently works with TNC to model the effects of climate change on TFFF and other TNC properties in the lower Altamaha River basin and to identify and restore TFFF in the watershed. A letter of support from TNC is included in appendix A.

References

An, S., and W.S. Gardner. 2002. Dissimilatory nitrate reduction to ammonium (DNRA) as a nitrogen link, versus denitrification as a sink in a shallow estuary (Laguna Madre/Baffin bay, Texas. *Marine Ecology – Progress Series* 237:41-50.

Anderson, C.J., and B.G. Lockaby. 2007. Chapter 3 – Soils and biogeochemistry of tidal freshwater floodplain wetlands. *In* W.H. Conner, T.W. Dole, and K.W. Krauss (ed.) Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States. Springer, New York. In press.

APHA (American Public Health Association), 1998. Standard methods for the examination of water and wastewater. American Public Health Association, Washington, DC.

Brinson, M.M. 1990. Riverine forests. Pp. 87-141. *In* A.E. Lugo, M.M. Brinson, and S.W. Brown (ed.), Forested Wetlands, Vol. 15. Ecosystems of the World. Elsevier, Science Publishers B.V., Amsterdam, The Netherlands.

Brinson, M.M., H.D. Bradshaw, R.N. Holmes, and J.B. Elkins, Jr. 1980. Litterfall, stemflow, and throughfall nutrient fluxes in an alluvial swamp forest. *Ecology* 61:827-835.

Brinson, M.M. 1984., H.D. Bradshaw and E.S. Kane. 1984. Nutrient assimilative capacity of an alluvial floodplain swamp. *Journal of Applied Ecology* 21:1041-1057.

Brinson, M.M., R.R. Christian, and L.K. Blum. 1995. Multiple states in the sea level induced transition from terrestrial forest to estuary. *Estuaries* 18:648-659.

Brunet, R.C., and L.J. Garcia-Gil. 1996. Sulfide-induced dissimilatory nitrate reduction to ammonia in anaerobic freshwater sediments. *FEMS Microbiology Ecology* 21:131-138.

Cahoon, DR. 1994. Recent accretion in two managed marsh impoundments in coastal Louisiana. *Ecological Applications* 4:166-176.

Caraco, N.F., J.J. Cole, and G.E. Likens. 1989. Evidence for sulphate-controlled phosphorus release from sediments of aquatic systems. *Nature* 341:395-413.

Church, J.A., J.M. Gregory and others. 2001. Chapter 11: Changes in sea level. p. 639-693. In J.T. Houghton and others (ed.), Climate Change 2001: The Scientific Basis. Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, UK.

Conner, W.H., and J.W. Day. 1982. The ecology of forested wetlands in the southeastern United States. P. 69-87. *In* B/ Gopal, R.E. Turner, R.G. Wetzel, and D.F. Whigham (ed.), Wetlands Ecology and Management. National Institute of Ecology and International Scientific Publications. New Delhi, India.

W.H. Conner, T.W. Dole, and K.W. Krauss (ed.). 2007. Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States. Springer, New York. In press.

Craft, C.B. 2007. Freshwater input structures soil properties, vertical accretion and nutrient accumulation of Georgia and United States (U.S.) tidal marshes. *Limnology and Oceanography* 52:1220-1230.

Craft, C.B., and W.P. Casey. 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. *Wetlands* 20:323-332.

Craft, C. and J. Schubauer-Berigan. 2006. The role of freshwater wetlands in a water quality trading program. P. 143-158. In, D. Wichelns (ed.) Proceedings: Innovations in Reducing Non-point Source Pollution: Methods, Policies, Programs, and Measurements. November 28-30, 2006. Indianapolis, IN.

Craft, C.B., S.W. Broome, and C.L. Campbell. 2002. Fifteen years of vegetation and soil development following brackish-water marsh creation. *Restoration Ecology* 10:248-258.

Cusbasch, U., G.A. Meehl and others. 2001. Chapter 9: Projections of future climate change. p. 525-582. In J.T. Houghton and others (ed.), Climate Change 2001: The Scientific Basis. Intergovernmental Panel on Climate Change, Cambridge University Press, Cambridge, UK.

Daily, G.C., S. Alexander, P.R. Ehrlich, L. Goulder, J. Lubchenco, P.A. Matson, H.A. Mooney, S. Postel, S.H. Schneider, D. Tilman, and G.M. Woodwell. 1997. Ecosystem services: Benefits supplied to human societies by natural ecosystems. *Issues in Ecology* 2:1-16.

Darke, A.K., and M.R. Walbridge. 2000. Al and Fe biogeochemistry in a floodplain forest: implications for P retention. *Biogeochemistry* 51:1-32.

De Groot, R.S., M.A. Wilson, and R.M.J. Boumans. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* 41:393-408.

DeLaune, R.D., W.H. Patrick, Jr., and S.R. Pezeshki. 1987. Foreseeable flooding and death of coastal wetland forests. *Environmental Conservation* 14:129-132.

Doyle, T.W., C.P. O'Neil, M.P.V. Melder, A.S. From, and M.M. Palta. 2007. Chapter 1 – Tidal freshwater swamps of the southeastern United States: Effects of land use, hurricanes, sea level rise, and climate change. *In* W.H. Conner, T.W. Dole, and K.W. Krauss (ed.) Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States. Springer, New York. In press.

Fowler, B.K., and C. Hershner. 1989. Primary production in Cohoke Swamp; a tidal freshwater wetland in Virginia. p. 365-374. *In* R.R. Sharitz and J.W. Gibbons (ed.) Freshwater Wetlands and Wildlife Symposium: Perspectives on Natural, Managed and Degraded Ecosystems. U.S. Department of Energy, Office of Scientific and Technical Information. Oak Ridge, TN. Conf-8603101, DOE Symposium series no. 61.

Frost, J.W., T. Schleicher, and C. Craft. Nitrogen limits primary and secondary production in a Georgia (USA) tidal freshwater marsh. Wetlands. In review.

Hackney, C.T. and G.F. Yelverton. 1990. Effects of human activities and sea level rise on wetland ecosystems in the Cape Fear River estuary, North Carolina. Pp. 55-61. In, D.F. Whigham et al. (ed.), Wetlands Ecology and management: Case Studies. Kluwer Academic Publishers. Dordrecht, The Netherlands.

Hanson, G.C., P.M. Goffman and A.J. Gold. 1994. Denitrifcation in riparian wetlands receiving high and low groundwater nitrate inputs. *Journal of Environmental Quality* 23:917-922.

Herrman, K.S., and J.R. White. Denitrification in intact sediment cores from a constructed wet-land: examining the isotope pairing technique. *Applied Geochemistry*. In review

Hickey, A.C., J.F. Kerestes, and B.E. McCallum. 2003. Water Resources Data – Georgia 2002. Volume 1: Continuous water level, streamflow, water quality data, and periodic water quality data, Water Year 2002. U.S. Geological Survey, Water Data report GA-02-01, Atlanta, Georgia.

Higgenbotham, C.B., M. ALber, and A.G. Chalmers. 2004. Analysis of tidal marsh vegetation patterns in two Georgia estuaries using aerial photography and GIS. *Estuaries* 27:670-683.

Howarth. R.W. 1988. Nutrient limitation of net primary production in marine ecosystems. *Annual review of Ecology and Systematics* 19:89-110.

Howarth, R.W., A. Sharpley, and D. Walker. 2002. Sources of nutrient pollution to coastal waters in the United Sates: Implications for achieving water quality goals. *Estuaries* 25:656-676.

Joye, S.B., and J.T. Hollibaugh. 1995. Influence of sulfide inhibition of nitrification on nitrogen regeneration in sediments. *Science* 270:623-625.

Karl, T.R., R.W. Knight, and N. Plummer. 1995. Trends in high-frequency climate variability in the 20th century. *Nature* 377:217-220.

Keller, J.K., J.R. White, S.D. Bridgham, and J. Pastor. 2004. Climate change effects on carbon and nitrogen mineralization in peatlands through changes in soil quality. *Global Change Biology* 10:1053-1064.

Kroes, D.E., C.R. Hupp, and G.B. Noe. 2007. Chapter 5 – Sediment, nutrient, and vegetation trends along the tidal, forested Pocomoke River, Maryland. *In* W.H. Conner, T.W. Dole, and K.W. Krauss (ed.) Ecology of Tidal Freshwater Forested Wetlands of the Southeastern United States. Springer, New York. In press.

Kuenzler, E.J., P.J. Mulholland, L.A. Yarbro, and L.A. Smock. 1980. Distributions and budgets of carbon, phosphorus, iron and manganese in a floodplain swamp ecosystem. Water Resources Research Institute of the University of North Carolina. Raleigh, NC, USA. Report No. 157.

Lamers, L.P.M., G. Dolle, S.T.G. Van Den Berg, S.P.J. Van Delft, P.J. Sebastiaan, and J.M. Roelofs. 2001. Differential responses of freshwater wetland soils to sulphate pollution. *Biogeochemistry* 55:87-102.

Lowrance, R., R. Leonard, L. Asmussen. 1985. Nutrient budgets for agricultural watersheds in the southeastern coastal plain. *Ecology* 66:2887-296.

Lowrance, R., R. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, L. Asmussen. 1984. Riparian forests and nutrient filters in agricultural watersheds. *BioScience* 34: 374-377.

Mahlman, J.D. 1997. Uncertainties in projections of human-caused climate warming. *Science* 278:1416-1417.

Mearns, L.O., F. Giorgi, L. McDaniel, and C. Shields. 2003. Climate scenarios for the southeastern U.S. based on GCM and regional model simulations. *Climatic Change* 60:7-35.

Meehl, G. A., Stocker, T. F., Collins, W. D., Friedlingstein, P., Gaye, A. T., Gregory, J. M., Kitoh, A., Knutti, R., Murphy, J. M., Noda, A., Raper, S. C. B., Watterson, I. G., Weaver, A. J. & Zhao, Z.-C. 2007. Global climate projections. Pp. 747-845. In, S. Solomon, D., Qin, M., Manning, Z., Chen, M., Marquis, K.B., Averyt, M., Tignor, and H.L. Miller, (eds.) Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge, United Kingdom and New York, NY.

Megonigal ,J.P., W.H. Conner, S. Kroeger, and R.R. Sharitz. 1997. Aboveground production in southeastern floodplain forests: a test of the subsidy-stress hypothesis. *Ecology* 78:370-384.

Moorhead, K.K., and M.M. Brinson. 1995. Response of wetlands to rising sea level in the lower coastal plain of North Carolina. *Ecological Applications* 5:261-271.

Noe, G.B., and C.R. Hupp. 2005. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic coastal plain rivers, USA. *Ecological Applications* 15:1178-1190.

Park, R.A. 1991. Global climate change and greenhouse emissions. p. 171-182. Subcommittee on Health and Environment, U.S. House of Representatives, Washington DC.

Park, R.A., T.V. Armentano, and C.L. Cloonan. 1986. Predicting the effects of sea level rise on coastal wetlands. p. 129-152. *In* J. G. Titus (ed.), Effects of Changes in Stratospheric Ozone and Global Climate, Vol. 4: Sea Level Rise. U.S. Environmental Protection Agency, Washington, D.C.

Park, R.A., J.K. Lee, P.W. Mausel, and R.C. Howe. 1991. Using remote sensing for modeling the impacts of sea level rise. *World Resources Review* 3:184-220.

Park, RA, MS Trehan, PW Mausel and RC Howe. 1989. Coastal wetlands in the twenty first century: Profound alterations due to rising sea level. Pp. 71-80. In, *Wetlands: Concerns and Successes*. Proceedings of a Symposium sponsored by American Water Resources Association, Bethesda, MD.

Pearlstine, L.G., W.M. Kitchens, P.J. Latham, and R.D. Bartleson. 1993. Tide gate influences on a tidal marsh. *Water Resource Bulletin* 29:1009-1019.

Pezeshki, S.R., R.D. Delaune, and W.H. Patrick, Jr. 1990. Flooding and saltwater intrusion: potential effects on survival and productivity of wetland forests along the U.S. Gulf Coast. *Forest Ecology and Management* 33/34:287-301.

Poe, A.C., M.F. Piehler, S.P. Thompson, and H.W. Paerl. 2003. Denitrification in a constructed wetland receiving agricultural runoff. *Wetlands* 23:817-826.

Rahmstorf, S., A. Cazenave, J.A. Church, J.E. Hansen, R.F. Keeling, D.E. Parker, and R.C.J. Somerville. 2007. Recent climate observations compared to projections. *Science* 316:709.

Rheinhardt, R. 1992. A multivariate analysis of vegetation patterns in tidal freshwater swamps of lower Chesapeake Bay, USA. *Bulletin of the Torrey Botanical Club* 119:192-207.

Rheinhardt, R.D., and C. Hershner. 1992. The relationship of belowground hydrology to canopy composition in five tidal freshwater swamps. *Wetlands* 12:208-216.

Richardson, CJ. 1994. Ecological functions and human values in wetlands: A framework for assessing forestry impacts. *Wetlands* 14:1-9.

Rysgaard, S., P. Thastum, T. Dalsgaard, P.B. Christensen, and N.L. Sloth. 1999. Effects of salinity on NH_4^+ adsorption capacity, nitrification and denitrification in Danish estuarine sediments. *Estuaries* 22:21-30.

SAS (Statistical Analysis Systems). 1996. SAS User's Guide. Cary, NC.

Sokal, R.R. and F.J. Rohlf. 1995. Biometry. W.H. Freeman and Co., San Francisco.

Steingruber, S.M., J. Friedrich, R. Gächter, and B. Wehrli. 2001. Measurement of denitrification in sediments with the ^{15}N isotope pairing technique. *Applied and Environmental Microbiology* 67:3771-3778.

Sundareshwar, P.V, and J.T. Morris. 1999. Phosphorus sorption characteristics of intertidal marsh sediments along an estuarine salinity gradient. *Limnology and Oceanography* 44: 1693-1701.

Titus, J. G., R. A. Park, S. P. Leatherman, J. R. Weggel, M. S. Greene, P. W. Mausel, M. S. Trehan, S. Brown, C. Grant, and G. W. Yohe. 1991. Greenhouse effect and sea level rise: Loss of land and the cost of holding back the sea. *Coastal Management* 19:171-204.

Thompson, S.P., M.F. Piehler, and H.W. Paerl. 2000. Denitrification in an estuarine headwater creek within an agricultural watershed. *Journal of Environmental Quality* 29:1914-1923.

Ulrich, G.A., L.R. Krumholz, and J.M. Suflita. 1997. A rapid and simple method for estimating sulfate reduction acitivity and quantifying inorganic sulfides. *Applied and Environmental Microbiology* 63:1627-1630.

Walbridge, M.R., and J.P. Struthers. 1993. Phosphorus retention in non-tidal palustrine forested wetlands of the mid-Atlantic region. *Wetlands* 13:84-94.

Waring, R.H., and W.H. Schlesinger. 1985. Forest Ecosystems: Concepts and Management. Academic Press, Inc., Orlando, 340 pp.

Weston, N.B., R.E. Dixon, and S.B. Joye. 2006. Ramifications of salinity intrusion in tidal freshwater sediments: Geochemistry and microbial pathways of organic matter mineralization. *Journal of Geophysical Research, Biogeosciences* 111, G0100, doi:10.1029/2005JG000071.

Wharton, C.H. 1978. The Natural Environments of Georgia. Georgia Department of Natural Resources, Atlanta, Georgia. 227 p.

Yarbro, L.A. 1983. The influence of hydrologic variations on phosphorus cycling and retention in a swamp stream ecosystem. Pp. 223-245. In D. Fontaine and S.M. Bartell (ed.), Dynamics of Lotic Ecosystems. Ann Arbor Science, Ann Arbor, Michigan.

Christopher B. Craft

Indiana University, School of Public and Environmental Affairs
1315 East 10th Street, Bloomington, IN 47405
(812) 855-5971 direct, (812) 855-7802 fax
e-mail: ccraft@indiana.edu
web: http://www.iu.edu/~speaweb/faculty/craft.php

a. Professional Preparation

University of North Carolina, Asheville	Biology	B.A.1980
University of Tennessee. Knoxville	Ecology	M.S. 1983
North Carolina State University, Raleigh	Soil Science	Ph.D.1987

b. Appointments

1999-present, Associate Professor, Indiana University, Bloomington IN
1997-1999, Assistant Scientist, Joseph W. Jones Ecological Research Center, Newton GA
1996-1997, Assistant Professor, University of Louisville, Louisville KY
1989-1995, Research Assistant Professor, Duke University, Durham NC
1987-1989, Research Associate, North Carolina State University. Raleigh, NC.

c. Publications

i) publications related to the proposed project

- Frost, J.W., T. Schleicher, and C. Craft. Nitrogen limits primary and secondary production in a Georgia (USA) tidal freshwater marsh. Wetlands. In review.
- Craft, C.B. 2007. Freshwater input structures soil properties, vertical accretion and nutrient accumulation of Georgia and United States (U.S.) tidal marshes. Limnology and Oceanography 52:1220-1230.
- Craft, C. and J. Schubauer-Berigan. 2006. The role of freshwater wetlands in a water quality trading program. P. 143-158. In, D. Wichelns (ed.) Proceedings: *Innovations in Reducing Nonpoint Source Pollution: Methods, Policies, Programs, and Measurements.* November 28-30, 2006. Indianapolis, IN.
- Aldous, A., P. McCormick, C. Ferguson, S. Graham and C. Craft 2005. Hydrologic regime controls soil phosphorus fluxes in restoration and undisturbed wetlands. Restoration Ecology 13:341-347.
- Craft, C.B. and W.P. Casey. 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. Wetlands 20:323-332.

ii) Other significant publications

- Graham, S.A., C.B. Craft, P. McCormick and A. Aldous. 2005. Forms and accumulation of soil P in natural and recently restored peatlands - Upper Klamath Lake, Oregon. Wetlands 25:594-606.

- Craft, C.B. and C. Chiang. 2002. Forms and amounts of soil nitrogen and phosphorus across a longleaf pine – depressional wetland landscape. Soil Science Society of America Journal 66:1713-1721.
- Craft, C.B., J.P. Megonigal, S.W. Broome, J. Cornell, R. Freese, R.J Stevenson, L. Zheng and J. Sacco. 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. Ecological Applications 13:1417-1432.
- Craft, C.B., S.W. Broome and C.L. Campbell. 2002. Fifteen years of vegetation and soil development following brackish-water marsh creation. Restoration Ecology 10:248-258.
- Craft, C.B. and C.J. Richardson. 1998. Recent and long-term organic soil accretion and nutrient accumulation in the Everglades. Soil Science Society of America Journal 62:834-843.

d. Synergistic Activities

- PI and Project Director. USEPA. Effects of Climate Change on Ecosystem Services of Tidal marshes and Coral Reefs-Science to Achieve Results (STAR) Program. Effects of sea level rise and climate variability of tidal marshes, South Atlantic Coast. $749,974. April 1 2005-March 31 2008.
- Co-PI. NSF. LTER GCE II: Georgia land-ocean margin ecosystem. November 15 2006-November 2012. $4,919,988.
- Vice President, Society of Wetland Scientists. June 2007-present.
- Chair, Division S-10 (Wetland Soils), Soil Science Society of America. 2003-2004.
- Associate Editor, *Wetlands*. 2007-present.
- Associate Editor, *Soil Science Society of America Journal.* 2004-present.
- Wetlands Expert Working Group: Developing Nutrient Criteria for Wetland Systems. US Environmental Protection Agency. Washington D.C. January 2000 - present.
- Member. Georgia Coastal Research Council. July 2006-present.
- Advisor. The Nature Conservancy (Altamaha and Ogeechee Rivers, Georgia). April 2006-present.
- Advisor. NOAA Restoration Center. July 2007-present.

JEFFREY R. WHITE

Biogeochemical Laboratories, Geology 301
Environmental Science, SPEA 300
Indiana University, Bloomington, IN 47405-2100
(812) 855-0731 E-mail: whitej@indiana.edu

a. Professional Preparation

Gettysburg College, Biology, B.A., 1977
Rutgers University, Environmental Science, M.S., 1979
Syracuse University, Civil Engineering, Ph.D., 1984

b. Appointments

Associate Vice Provost for Research, Indiana University, 2006-present
Associate Dean, Bloomington Programs, SPEA, Indiana University, 2001-2005
Professor, School of Public and Environmental Affairs and Department of Geological Sciences, Indiana University, 1998 - present
Chair, Environmental Science and Policy, School of Public and Environmental Affairs, Indiana University, 1994-1997
Visiting Research Professor, Marine Science, University of North Carolina, Chapel Hill, 1993-1994
Director, Ph.D. Program in Environmental Science, Indiana University, 1991-1993
Associate Professor, School of Public and Environmental Affairs and Department of Geological Sciences, Indiana University, 1990-1998
Assistant Professor, School of Public and Environmental Affairs, Indiana University, 1984-1990

c. (i) 5 Publications Most Closely Related to Project

Keller, J.K., White, J.R., Bridgham, S.D., Pastor, J. "Climate Change Effects on Carbon and Nitrogen Mineralization in Peatlands through Changes in Soil Quality." *Global Change Biology*, Vol. 10, 2004, pp. 1053-1064.
Avery, B., Shannon, R.D., White, J.R., Martens, C.S., and Alperin, M.J. "Controls on Methane Production in a Tidal Freshwater Estuary and a Peatland: Methane Production via Acetate Fermentation and CO_2 Reduction," *Biogeochemistry,* Vol. 62, 2002, pp. 19-37.
Avery, B., Shannon, R.D., White, J.R., Martens, C.S., and Alperin, M.J. "Effect of Seasonal Changes in the Pathways of Methanogenesis on the $\delta^{13}C$ Values of Pore Water Methane in a Michigan Peatland," *Global Biogeochemical Cycles*, Vol. 13, 1999, pp. 475-484.
Walter, B.P., Heimann, M., Shannon, R.D., White, J.R., "A process-based model to derive methane emissions from natural wetlands," *Geophysical Research Letters*, Vol. 23, 1996, 3731-3734.
Shannon, R.D., and White, J.R., "The Effects of Spatial and Temporal Variations in Acetate and Sulfate on Methane Cycling in Two Michigan Peatlands," *Limnology and Oceanography*, Vol. 41, 1996, pp. 435-443.

c. (ii) 5 Other Related Publications

Shannon, R.D., and White, J.R., "A Three-Year Study of Controls on Methane Emissions From Two Michigan Peatlands," *Biogeochemistry*, Vol. 27, 1994, pp. 35-60.

White, J.R., and Shannon, R.D., "Modeling Organic Solutes in Peatland Soils Using Acid Analogs," *Soil Science Society of America Journal*, Vol. 61, 1997, pp. 1257-1263.

Shannon, R.D., and White, J.R., "The Selectivity of a Sequential Extraction Procedure for Iron Oxyhydroxide and Sulfides in Freshwater Sediments," *Biogeochemistry*, Vol. 14, 1991, pp. 193-208.

Gubala, C.P., Engstrom, D.R. and White, J.R., "Effects of Iron Cycling on ^{210}Pb Dating of Sediments in an Adirondack Lake, U.S.A." *Canadian Journal of Fisheries and Aquatic Sciences*, Vol. 47, 1990, pp. 1821-1829.

White, J.R., Gubala, C.P., Fry, B., Owen, J., and Mitchell, M.J., "Sediment Biogeochemistry of Iron and Sulfur in an Acidic Lake", *Geochimica et Cosmochimica Acta*, Vol. 53, 1989, pp. 2547-2559.

d. Synergistic Activities

Invited Participant, Workshop on Quantification of CH_4 Emissions from Land Ecosystems: Integrating Field and In-situ Observations, Satellite Data, and Modeling, National Center for Ecological Analysis and Synthesis, Santa Barbara, CA, March 2006 – December 2009.

Member, Liaison Committee, National Water Quality Assessment Program, US Geological Survey, 1991-1999.

Gubernatorial Appointee, Indiana Interagency Watershed Task Force, 1992-1993.

Technical Consultant, National Acidic Precipitation Assessment Program (NAPAP), Environmental Protection Agency, 1985-1988.

Member, Funding Review Panels for NSF, EPA, DOE, USGS, Water Resource Research Institutes.

e. Collaborators & Other Affiliations

(i) Collaborators. Marc Alperin, Univ. North Carolina; Brooks Avery, Univ. North Carolina; Scott Bridgham, Univ. Notre Dame; Jiquan Chen, Univ. Toledo; Chris Craft, Indiana University; Jason Keller, Univ. Notre Dame; Chris Martens, Univ. North Carolina; John Pastor, Univ. Minnesota; Flynn Picardal, Indiana Univ.; Lisa Pratt; Indiana Univ.; Rob Shannon, Penn State Univ.; Jake Weltzin, Univ. Tennessee.

(ii) Thesis Advisor: Charles Driscoll, Syracuse University

(iii) Thesis Advisees: Todd Bish, Env. Consult., Pennsylvania; Ana Amelia Boischio, Nat. Univ. Brazil; Brad Gilmour, Env. Law, Calgary; Chad Gubala, Univ. Toronto; Nancy Hearne, EPA; Kyle Herrman, Ohio State Univ.; Joan Lawson, Ohio; Robert Shannon, Penn State Univ. (total of 8).
Postdoctoral sponsorships: Evelyn Krull, CSIRO, Adelaide, Australia; Robert Shannon, Penn State Univ. (total of 2)

Craig A. Wayson

Indiana University, School of Public and Environmental Affairs
1315 East 10th Street, Bloomington, IN 47405
(812) 855-4953, (812) 855-7547 fax
e-mail: cwayson@indiana.edu

a. Professional Preparation

Iowa State University, Ames	Animal Ecology, Environmental Studies	B.S. 1992
Indiana University, Bloomington	Environmental Science	M.S.E.S. 2002
Indiana University, Bloomington	Public Affairs	M.P.A. 2002
Indiana University, Bloomington	Environmental Science	Ph.D. 2005

b. Appointments

2005-present, Post-Doctoral Fellow, Indiana University, Bloomington IN

c. Publications

- Wayson, C.A., J.C. Randolph, P.J. Hanson, H.P. Schmid and C.S.B. Grimmond. 2005. Comparison of soil respiration methods in a mid-latitude deciduous forest. *Biogeochemistry* **80**:173-189.
- Oliphant A.J, S.B Grimmond; H.P. Schmid, C.A. Wayson. 2006. Local-scale heterogeneity of photosynthetically active radiation (PAR), absorbed PAR and net radiation as a function of topography, sky conditions and leaf area index. *Remote Sensing of the Environment* **103**:324-337.
- Ehman, J.L., H.P. Schmid, C.S.B. Grimmond, J.C. Randolph, P.J. Hanson, C.A. Wayson and F.D. Cropley. 2002. An initial intercomparison of micrometeorological and ecological inventory estimates of carbon exchange in a mid-latitude deciduous forest. *Global Change Biology*, **8**:575-589.

Current & Pending Support: Christopher B. Craft

Current Grants (for collaborative projects, award amount reflects Craft's portion of the total)

1. US EPA: Effects of Sea level Rise and Climate Variability on Ecosystem Services of Tidal Marshes, South Atlantic Coast.

   Source: EPA
   Award: $749,974
   Location: Indiana University
   Commitment: 1 summer mo.

   Period: 4/1/2006 – 3/31/2008

2. US DOI (Everglades National Park): Radiometric dating of wetland soil cores to re-construct historical water levels and plant communities of the Florida Everglades.

   Source: Florida International University
   Award: $10,106
   Location: Indiana University
   Commitment: .25 summer mo.

   Period: 1/1/2006 – 12/31/2007

3. NOAA (National Estuarine Research Reserve): Characterization of passerine food source, trophic structure and habitat utilization on Sapelo Island Georgia using stable isotopes of C, N and H.

   Source: NOAA
   Award: $60,000
   Location: Indiana University
   Commitment: 0
   Period: 5/1/2006 – 4/30/2009

4. NSF. Georgia Coastal Ecosystems Long Term Ecological Research. $4,919,998

   Source: University of Georgia
   Award: $91,391
   Location: Indiana University
   Commitment: 0.5
   Period: 11/15/2009 – 11/14/2013

Current & Pending Support: Christopher B. Craft

Pending Proposals

1. US DOI (Everglades National Park): Peat accretion in mangrove-sawgrass ecotones.

   Source: Florida International University
   Award: $9,610
   Location: Indiana University
   Commitment: .25 summer mo
   Period: 7/1/2007 – 6/30/2009

2. The Coastal Center of the U.S. Department of Energy (DOE) National Institute for Climatic Change Research (NICCR)

   Source: Department of Energy
   Award: $343,181
   Location: Indiana University
   Commitment: 1.0
   Period: 4/1/08– 3/31/2012

Current & Pending Support: Jeffrey R. White

Pending Proposals

1. DOE – Ecosystem-Atmosphere Exchange of Carbon, Water and Energy over a Mixed Deciduous Forest in the Midwest

Source:	DOE
Award:	$236,893
Location:	Indiana University
Commitment:	0.5
Period:	9/1/2007 – 8/31/2010

Annotated Bibliography of Prior NIGEC Research

PI Craft: No previous support

Co-PI Wayson: No previous support

Co-PI White:

Grants funded:

"An Investigation of the Physical and Biogeochemical Processes Controlling Methane Emissions From Peatland Ecosystems," National Institute for Global Environmental Change, U.S. Department of Energy, Midwestern Regional Center, Indiana University, July 1993-June 1994, $65,000; Principal Investigator, Jeffrey R. White, Indiana University; Co-principal Investigator, Robert D. Shannon, Indiana University.

"Temporal and Spatial Variability of Methane Cycling in Wetland Ecosystems of the Northern Temperate Zone," National Institute for Global Environmental Change, U.S. Department of Energy, Midwestern Regional Center, Indiana University, July 1992-June 1993, $149,000; Principal Investigator, Jeffrey R. White, Indiana University; Co-principal Investigator, Robert D. Shannon, Indiana University.

"Temporal and Spatial Variability of Methane Cycling in Wetland Ecosystems of the Northern Temperate Zone," National Institute for Global Environmental Change, U.S. Department of Energy, Midwestern Regional Center, Indiana University, July 1991-June 1992, $156,000; Principal Investigator, Jeffrey R. White, Indiana University.

"Temporal and Spatial Variability in Methane Emissions From Wetland Ecosystems in the Northern Temperate Zone," National Institute for Global Environmental Change, U.S. Department of Energy, Midwestern Regional Center, Indiana University, August 1990-July 1991, $110,500; Principal Investigator, Jeffrey R. White, Indiana University.

Publications Produced:

Avery, B., Shannon, R.D., White, J.R., Martens, C.S., and Alperin, M.J. "Controls on Methane Production in a Tidal Freshwater Estuary and a Peatland: Methane Production via Acetate Fermentation and CO_2 Reduction," *Biogeochemistry*, Vol. 62, 2002, pp. 19-37.

Avery, B., Shannon, R.D., White, J.R., Martens, C.S., and Alperin, M.J. "Effect of Seasonal Changes in the Pathways of Methanogenesis on the $\delta^{13}C$ Values of Pore Water Methane in a Michigan Peatland," *Global Biogeochemical Cycles*, Vol. 13, 1999, pp. 475-484.

Walter, B.P., Heimann, M., Shannon, R.D., White, J.R., "A process-based model to derive methane emissions from natural wetlands," *Geophysical Research Letters*, Vol. 23, 1996, 3731-3734.

Shannon, R.D., and White, J.R., "The Effects of Spatial and Temporal Variations in Acetate and Sulfate on Methane Cycling in Two Michigan Peatlands," *Limnology and Oceanography*, Vol. 41, 1996, pp. 435-443.

Shannon, R.D., and White, J.R., "A Three-Year Study of Controls on Methane Emissions From Two Michigan Peatlands," *Biogeochemistry*, Vol. 27, 1994, pp. 35-60.

White, J.R., and Shannon, R.D., "Modeling Organic Solutes in Peatland Soils Using Acid Analogs," *Soil Science Society of America Journal*, Vol. 61, 1997, pp. 1257-1263.

Eco Modeling

July 2, 2007
Re: **NICCR RFP-03**

DOE National Institute for Climatic Change Research

Subject proposal: Effects of Accelerated Sea Level Rise and Variable Freshwater River Discharge on Water Quality Improvement Functions of Tidal Freshwater Floodplain Forests

To whom it may concern:

I am writing in support of Professor Christopher Craft's proposal. I have worked closely with him on the US EPA-funded project on "Effect of Sea Level Rise and Climate Variability." He and his associate Jeff Ehman have been instrumental in the improvement and implementation of our SLAMM sea-level rise model. The project is noteworthy in that it has incorporated a salinity algorithm so that the effects of both sea-level rise and changes in discharge of coastal rivers can be simulated.

Prof. Craft's knowledge of the southeastern estuaries has been invaluable in development of SLAMM 5 in this ongoing project. His research on wetland ecosystem services provides the scientific basis for predicting futue impacts. Although the model includes forested wetlands, they are not the thrust of the current project. The proposed project would extend this work and would result in a more general model of river-dominated coastal ecosystems subject to climate change.

Sincerely,

Richard A. Park, Ph.D.
President

5522 Alakoko Place
Diamondhead MS 39525

(228) 255-9841
fax (228) 255-1496

dickpark@CableOne.net
http://www.myweb.cable one.net/dickpark/

The Nature Conservancy
SAVING THE LAST GREAT PLACES ON EARTH

The Nature Conservancy
Southeast Georgia Conservation Office
P.O. Box 484
Darien, GA 31305-0484

tel [912] 437.2161
fax [912] 437.5368
nature.org

27 November 2006

Department of Energy, National Institute for Climatic Change Research

RE: The Grant Proposal, *Effects of Accelerated Sea Level Rise and Variable Freshwater River Discharge on Ecosystem Services of Freshwater Tidal Floodplain Forests*

To Whom It May Concern:

The Nature Conservancy has been working to protect the Altamaha River watershed for over 30 years. Beginning in 1969 with the acquisition of Wolf and Egg islands, which are now managed by the U.S. Fish and Wildlife Service as part of the Savannah National Wildlife Refuge, our organization has worked with private, public and corporate landowners and conservation partners to ensure the permanent protection of over 79,000 acres of land along the Altamaha River. In 1991, The Nature Conservancy designated the Altamaha River as one of the world's 75 Last Great Places and established the Altamaha River Bioreserve, a community-based project solely dedicated to the conservation of the river and surrounding lands. As we continue to work with communities and partners to achieve our mission, the Altamaha River promises to be a conservation priority for The Nature Conservancy over the next several decades.

Long recognized for its historical, cultural and natural values, the Altamaha River's natural communities provide critical habitat and food sources for the survival of a wide array of plants and animals, including the largest documented aggregation of globally imperiled plants and animals of any watershed in Georgia. Many of these plants, animals and natural communities (such as small stream swamps) contribute to the economic health of the surrounding region and Georgia as a whole, helping to support a multimillion-dollar commercial and recreational fishing and tourism industry. Thus, the economic stability of the region and ecological well being of the Altamaha River and its tributaries are closely linked and interdependent.

The Nature Conservancy supports Christopher Craft development of scientific research on the tidal and non-tidal floodplain swamps of the lower Altamaha River. His research of NPP, water quality and migratory birds benefit a number of our conservation targets. The Nature Conservancy has partnered with Christopher Craft, Georgia Department of Natural Resources - Coastal Resources and Wildlife Resources divisions, US Fish and Wildlife Services and Sapelo Island National Estuarine Research Reserve to cooperatively identify and restore tidal floodplain forests. The work described in the grant proposal *Effects of Accelerated Sea Level Rise and Variable Freshwater River Discharge on Ecosystem Services of Freshwater Tidal Floodplain Forests* will help document some of the important ecosystem services these wetlands provide and allow for scienctically prepared restoration and monitoring plans. The Nature Conservancy is dedicated to efforts of our partners for adaptive conservation actions including forest protection and restoration projects to anticipate climate change impacts.

The Nature Conservancy fully supports the grant proposal, *Effects of Accelerated Sea Level Rise and Variable Freshwater River Discharge on Ecosystem Services of Freshwater Tidal Floodplain Forests*, to the Department of Energy, National Institute for Climatic Change Research.

Sincerely,

Christi Lambert

Christi Lambert
Southeast Georgia Conservation Director

Jeffrey Spratt
Conservation Assistant

OMB Approval No. 0348-0043

APPLICATION FOR FEDERAL ASSISTANCE

2. DATE SUBMITTED 1/10/06	Applicant Identifier DUNS 00-604-6700
3. DATE RECEIVED BY STATE	State Applicant Identifier
4. DATE RECEIVED BY FEDERAL AGENCY	Federal Identifier

1. TYPE OF SUBMISSION

Application
- ☐ Construction
- ☒ Non-Construction

Preapplication
- ☐ Construction
- ☐ Non-Construction

5. APPLICANT INFORMATION IS THIS PROPOSAL BEING SUBMITTED TO ANOTHER FEDERAL AGENCY? ☐YES ☒NO IF YES, LIST ACRONYM(S)

Legal Name:
Trustees of Indiana University

Organizational Unit:
School of Public and Environmental Affairs

Address *(give city, county, state, and zip code)*:
P.O. Box 1847
Bloomington, IN 47402-1847
Monroe County

Name and telephone and E-mail number of the person to be contacted on matters involving this application *(give area code)*
PI: Christopher Craft, 812-855-5971 ccraft@indiana.edu

ADMIN. CONTACT: Teresa Miller, 812-855-0516 RUGS@indiana.edu

6. EMPLOYER IDENTIFICATION NUMBER (EIN): 35 — 6001673

7. TYPE OF APPLICANT: *(enter appropriate letter in box)* I

A. State	H. Independent School Dist.
B. County	I. State Controlled Institution of Higher Learning
C. Municipal	J. Private University
D. Township	K. Indian Tribe
E. Interstate	L. Individual
F. Intermunicipal	M. Profit Organization
G. Special District	N. Other (Specify)

8. TYPE OF APPLICATION:

☒ New ☐ Continuation ☐ Revision

If Revision, enter appropriate letter(s) in box(es):

A. Increase Award B. Decrease Award C. Increase Duration
D. Decrease Duration Other *(specify)*:

9. NAME OF FEDERAL AGENCY:

U.S. Environmental Protection Agency - ORD - NCER

10. CATALOG OF FEDERAL DOMESTIC ASSISTANCE NUMBER: __ __ . __ __ __

TITLE: Watershed Restoration for the Hatfield Restoration Program

11. DESCRIPTIVE TITLE OF APPLICANT'S PROJECT:
Identifying Watershed Sources of Phosphorus to Upper Klamath Lake to Achieve TMDL Compliance

12. AREAS AFFECTED BY PROJECT *(cities, counties, states, etc.)*:
Oregon and California

13. PROPOSED PROJECT:

Start Date	Ending Date
7/1/2006	6/30/2008

14. CONGRESSIONAL DISTRICTS OF:

a. Applicant	b. Project
9th	Indiana, 9h

15. ESTIMATED TOTAL PROJECT FUNDING:

a. Federal	$	49,930.00
b. Applicant	$	17,075.00
c. State	$	.00
d. Local	$	.00
e. Other	$	.00
f. Program Income	$	.00
g. TOTAL	$	67,006..00

16. IS APPLICATION SUBJECT TO REVIEW BY STATE EXECUTIVE ORDER 12372 PROCESS?

a. YES. THIS PREAPPLICATION/APPLICATION WAS MADE AVAILABLE TO THE STATE EXECUTIVE ORDER 12372 PROCESS FOR REVIEW ON:

DATE ____________

b. NO. ☐ PROGRAM IS NOT COVERED BY E.O. 12372

☒ OR PROGRAM HAS NOT BEEN SELECTED BY STATE FOR REVIEW

17. IS THE APPLICANT DELINQUENT ON ANY FEDERAL DEBT?

☐ Yes If "Yes," attach an explanation. ☒ No

18. TO THE BEST OF MY KNOWLEDGE AND BELIEF, ALL DATA IN THIS APPLICATION/PREAPPLICATION ARE TRUE AND CORRECT. THE DOCUMENT HAS BEEN DULY AUTHORIZED BY THE GOVERNING BODY OF THE APPLICANT AND THE APPLICANT WILL COMPLY WITH THE ATTACHED ASSURANCES IF THE ASSISTANCE IS AWARDED.

a. Typed Name of Authorized Representative	b. Title	c. Telephone number
Michael A. McRobbie	Vice President for Research	812-855-0516

d. Signature of Authorized Representative	e. Date Signed
Michael A McRobbie	1.11.06

APPENDIX 5 HATFIELD RESTORATION PROGRAM FISCAL YEAR 2006

PROJECT SUMMARY SHEET

PROJECT TITLE: **Identifying Watershed Sources of Phosphorus to Upper Klamath Lake to Achieve TMDL Compliance**

PROPOSER/ORGANIZATION: **The Trustees of Indiana University**
ADDRESS: **P.O. BOX 1847**
CITY: **Bloomington**
STATE: **IN**
ZIP CODE: **47402-1847**

ADMINISTRATIVE CONTACT PERSON: **Teresa Miller, Director of Sponsored Program**
TELEPHONE NUMBER: **812-855-0516 office, 812-855-9943 fax**

PROJECT OBJECTIVE: **Determine the effects of land use (crop, pasture, sage brush, forest), geomorphic position (stream bank, stream bed) and soil type on soil erosion and particulate P transport to the Sprague River and UKL.**

FUNDING REQUESTED: **84,907**

COST SHARE FUNDS or IN-KIND CONTRIBUTIONS: **34,151**

LOCATION (Sub-basin; USGS Quad; Township, Range, Section)
Sprague River Valley

PROJECT DESCRIPTION AND SPECIES BENEFITED: **Soil erosion/deposition and particulate P transport will be measured under different land use (crop, pasture, sage brush, forest), landscape position (stream bank, stream bed) and soil types of the Sprague River Valley. Total, organic and bioavailable P will be measured in surface soils and sediment. The inventory of ^{137}Cs in soil will be used to estimate soil and P erosion from uplands and sediment and P deposition in the stream bed. Lab incubations will be performed to gauge P release from soils/sediments under aerobic/anaerobic (sediments) conditions. This work is important because it will provide an estimate of the role of soil erosion and particulate P transport to the Sprague River and UKL.**

PARTNERS/COOPERATORS:

Project Title: Identifying Watershed Sources of Phosphorus to Upper Klamath Lake to Achieve TMDL Compliance

Project Proposers

Chris Craft, Indiana University, School of Public and Environmental Affairs, 1315 East 10th St, Bloomington, IN 47405, Ph: 812-855-7802, email: ccraft@indiana.edu

Paul McCormick, USGS, Leetown Science Center, 11649 Leetown Rd, Kearneysville, WV 25430 Ph: 304-724-4478, email: pmccormick@usgs.gov

Program Information

Improving water quality in Upper Klamath Lake (UKL) is a cornerstone of restoration in the Upper Klamath Basin and a critical element in the recovery plan for endangered fish populations in the lake. Current plans to improve water quality (ODEQ 2002) require reductions in watershed phosphorus (P) loading in order to meet required P concentrations in UKL. It is believed that non-point loading may contribute substantial P to the lake. However, the distribution and importance of such P sources within the watershed continues to be debated. Resolving this issue is a prerequisite to the development of effective watershed P control strategies and is critical for identifying Best Management Practices (BMP's) to improve water quality of UKL as outlined in the 5-year plan for restoration of Upper Klamath Basin. The proposed work addresses two *KBERO's 5-year Research and Assessment Needs:*
(1) assess "bench scale P" in SRV by quantifying P sources by land use and soil type in SRV. and *(2)* "Sprague River sediment bed load study" by identifying sources and locations of bed load.

Background

Phosphorus is the primary nutrient that leads to degradation of water quality of freshwater ecosystems. Excess P has been linked to water quality problems in UKL, including blooms of the algae *Aphanizomenon flos-aquae,* that degrade the habitat of two federally endangered fish, the Lost River sucker (*Deltistes luxatus*) and the shortnose sucker (*Chasmistes brevirostris*) (NRC 2004). Even though P is linked to water quality degradation of the lake, the source(s) of P to the lake remain poorly understood. Anthropogenic activities including cultivation, grazing and wetland drainage are thought to contribute P to the lake (Bortleson and Fretwell 1993) though natural sources such spring water containing high levels of P also have been implicated (NRC 2004). Of unknown importance are inputs of particulate P carried into the lake by eroded soils and sediments. Analysis of magnetic properties of UKL sediments suggest that deposition of "fresh" (recent) sediment to the lake increased beginning in the 1920's and continuing up to the present (Bradbury et al. 2004). It is hypothesized that drained wetlands AND other cleared land clearing in the Sprague River Valley (SRV) and the Williamson River catchment are the source of this material (Bradbury et al. 2004). It is not known, though, whether high P concentrations in the lake are linked to increased sediment transport coupled to soil erosion.

Human activities such as tillage and grazing promote soil erosion. On cultivated lands, eroded soil particles account for 50% of P leaving the field, mostly in association with clay-size particles (Cooper and Gilliam 1987). Grazing also promotes erosion, especially on hill slopes and stream banks where foot traffic is high. Soils within the Klamath Basin are highly erodible when disturbed (USFWS 1998) and recent paleolimnological work in the lake provides evidence that erosion rates within the watershed have increased dramatically during the past century (Eilers et al. 2001). Accelerated upland soil and stream bank erosion caused by the conversion of native sage and forest habitat to irrigated pasture and cropland may represent the major human contribution to P loading in the UKL watershed, especially in the SRV. We will evaluate this hypothesis that the primary human influence on P dynamics in the UKL watershed has been to increase the rate of transport of native P downstream, primarily in particulate form.

Assessing the contribution of soil erosion and sediment transport to lake eutrophication requires an understanding not only of the quantity of sediment P loads but of the potential availability of this P for algal growth. Typically, <60% of sediment TP consists of biologically available P (BAP) (Sonzogni et al. 1982), and this amount can vary among sediment sources (Logan et al. 1979). Thus, measures of soil BAP provide a better indication of a source soil's potential to contribute to downstream eutrophication than the TP content alone (Sharpley et al. 1995).

Quantifying nonpoint source loading from different sources can be costly and problematic. For example, accurate estimates of sediment P loading from different sources to UKL will require a long-term commitment to spatially intensive and well-timed water quality monitoring, as most sediment transport undoubtedly occurs during large but infrequent flood events. Instead, we propose a rapid assessment of the potential contribution of different upland and riparian/stream bank sediment sources to total P loads and the influence of soil characteristics and in-stream and lake processes on the bioavailability of sediment P from these sources. The goal of our work is to isolate the most likely sources of bioavailable P so that limited restoration funds can be focused on projects that can maximize P load reductions to UKL.

Project Objectives

The objectives of this study are to determine:

(1) Whether erosion of upland and stream bank soils in the SRV constitutes a significant potential source of biologically available P (BAP) to UKL

(2) Whether agricultural (crop, grazing) lands exhibit higher levels of soil BAP and/or contribute a disproportionate amount of sediment to the river compared with native forest and sagebrush habitats

(3) Whether sediments accumulating in the stream bed between major storm events represent a significant pool of potential BAP for UKL

(4) What fraction of sediment P is actually bioavailable once these sediments have entered UKL, which experiences alternating periods of well-mixed (aerobic) and quiescent (reducing) conditions.

Tasks

Soils will be collected from different land uses (crop, pasture, sagebrush, forest), soil types and geomorphic positions (stream bank, streambed) and analyzed for BAP and total P to quantify labile and recalcitrant pools of P. Inventories of ^{137}Cs in soil will be measured to quantify rates of soil erosion exposed to different land uses as well as sediment deposition in the streambed. The ^{137}Cs and P data will be combined to estimate particulate P transport from uplands, including stream bank to the stream bed. Incubations of soils and sediments will be performed under aerobic and anaerobic conditions to determine release of BAP and its contribution to the P load to UKL. The proposed work addresses two *KBERO's 5-year Research and Assessment Needs: (1)* assess "Bench scale P" in SRV by quantifying P sources by land use and soil type in SRV. and *(2)* Sprague River sediment bed load study by identifying sources and locations of bed load.

Methods

Field Sampling

Four land uses (crop, pasture, sage, forest) and geomorphic positions (stream bank, stream bed) will be selected in the SRV. For each land use, two soil types representing the dominant (in terms of acreage) soils for a particular land use will be sampled (Table 1). Soils of the SRV are classified mostly as Mollisols and Alfisols (soils formed under grassland and scrub vegetation) though Entisols and Inceptisols are present on steeply sloping lands (Table 1). Soils will be collected from ten replicates of each land use and soil type (80 locations). Soils (10 replicates) also will be collected from each geomorphic position for a total of 20 locations. At each location, surface soils (0-5 cm) will be collected in triplicate, combined for analysis and shipped to the lab in Indiana. We chose surface soils (0-5 cm) because they represent potentially erodible soil material that is available for transport to the Sprague River.

Available soil and land-use maps for the SRV and consultation with other agencies working in the SRV (e.g., the Klamath Basin Ecosystem Restoration Office, National Resources Conservation Service, Oregon Department of Environmental Quality) will be used identify appropriate and accessible sampling locations and to finalize the sampling design. The goal will be to focus sampling effort on soil types most susceptible to erosion based on proximity to the SRV and its tributaries and to collect replicate samples of each type from locations having different land uses (e.g., native scrub or other minimally affected locations vs. grazing vs. crop production). This design will allow us to statistically determine the potential contribution of different soil types to downstream P loads and the effects of land use changes on this contribution.

Table 1. Land uses, soil types and taxonomic classification of soils that will be sampled in SRV.

Land Use	Soil Type	USDA Soil Taxonomy [1]
Crop	Klamath	fine, montmorillonitic Cumulic Cryaquolls
	Ontko	medial over loamy, mixed, nonacid Andic Cryaquept
Pasture	Klamath	fine, montmorillonitic Cumulic Cryaquolls
	Ontko	medial over loamy, mixed, nonacid Andic Cryaquept
Sage brush	Choptie	loamy, mixed, frigid Lithic Haploxeroll
	Yainax	fine-loamy, mixed, frigid Mollic Haploxeralf
Forest	Maset	loamy-skeletal, mixed nonacid Andeptic Cryorthent
	Woodcock	loamy-skeletal, mixed, argic Pachic Cryoboroll
Stream bank	NA	NA
Stream bed	NA	NA

[1] USDA (1985)
NA = not applicable

Soil Analyses

Soils will be air-dried, weighed and analyzed for BAP, organic P and total P. BAP will be extracted with 0.5 M H_2SO_4 and analyzed for phosphate-P (Kuo 1996). Organic P will be analyzed as phosphate-P after ignition at 550^oC (Kuo 1996). Total P will be measured as phosphate-P after digestion in nitric-perchloric acid (Sommers and Nelson 1972).

Supplemental analysis of soils will include measurements of bulk density, organic C, total N and particle size. Organic P in soil is positively linked to organic C content. P bound to mineral soil, through sorption and precipitation, is related to particle size, especially clay content (Cooper and Gilliam 1987). Bulk density will be measured by drying the soils at 105^oC, weighing them and dividing by the core volume. Organic C and total N will be measured using Perkin-Elmer CHN analyzer. Particle size will be determined using the hydrometer method (Gee and Or 2002).

<u>BAP, Organic P, Total P, Bulk Density, Organic C, N, Particle Size:</u>

((4 land uses x 2 soil types) + 2 geomorphic positions) x 10 replicates = 100 samples

A second set of soils will be collected from each land use and geomorphic position for analysis of ^{137}Cs. The inventory of ^{137}Cs in soils and sediments, when compared with cumulative atmospheric deposition of ^{137}Cs, can be used to estimate rates of erosion on hill slopes and sediment accumulation in depositional areas (Brown et al. 1981). Soils for ^{137}Cs analysis will be collected from five of the ten replicates of each land use and geomorphic position. Several (3-5) cores will be collected from undisturbed ridge top locations to determined ^{137}Cs inventories of undisturbed, uneroded soils. This technique was successfully used by Brown et al. (1981) to estimate recent erosion and sedimentation in the hilly margin of the Willamette Valley, Oregon.

Soils will be sampled in 15 cm increments to a depth of 45 cm and analyzed for ^{137}Cs by gamma analysis of the 661.62 keV photopeak (Graham et al. 2005). Cs-137 inventories will be calculated by multiplying ^{137}Cs activities by bulk density then summing up by inventories over the three depths.

<u>Cs-137:</u>

((4 land uses x 2 soil types) + 2 geomorphic positions) x 5 replicates x 3 soil depths = 150 samples

Lab Incubations

Continued processing of eroded soils that become part of the river and lake sediment pools can result in the release of additional P beyond that considered to be biologically available in the upland soil analyses. Phosphorus release from the sediment organic fraction is controlled by both the quantity and quality of the entrained organic matter (carbon) as well as other environmental conditions, particularly the availability of oxygen for microbial metabolism. Release of some metal-bound P occurs when sediments are exposed to anaerobic conditions.

In the laboratory, sediments from selected sampling locations in the SRV and its tributaries will be incubated in a water-saturated environment at near-ambient temperatures under either aerobic or anaerobic conditions for 4 weeks. Replicate samples will be harvested each week, and dissolved P will be extracted using a dilute $CaCl_2$ solution. The relationship between P loss rates and sediment conditions (% organic matter, P fractions, and aerobic vs. anaerobic conditions) will be evaluated using these data. Rates of P loss measured in this manner will be used to identify longitudinal (upstream-downstream) patterns in microbially mediated P mineralization rates and the potential significance of this process as a source of bioavailable P to the lake.

A second incubation will be performed using a representative subset of the sediment samples to assess the importance of C limitation on P mineralization rates. Microbial growth is often C limited in the soil/sediment environment and, consequently, the availability of C substrates may control the rate of P mineralization. Replicate samples will be incubated in the presence and absence of a labile C source (e.g., acetate), and the rate of P mineralization will be measured as described above. Bacterial biomass will be measured at the end of the incubation using the fumigation-extraction method (Vance et al. 1987) as it is possible that increased microbial growth may immobilize any additional P mineralized in the C-amended treatment. Alkaline phosphatase activity will be measured to assess whether a shift towards increasingly P limited growth occurs under conditions of high C availability. An understanding of the potential for P limited microbial growth is important to predicting sediment-P mineralization rates as this process is mediated by enzymatic pathways induced by P starvation.

Lab Incubations:

((4 land uses x 2 soil types) + 2 geomorphic positions) x 10 replicates = 100 samples

Statistical Analysis

Differences in soil P, erosion-deposition, supplemental soil properties and P release among land uses and soil types will be tested using analysis of variance (ANOVA). Univariate statistics will be performed first to ensure that it is normally distributed and that it estimates a common (homogeneity) variance in order to meet the assumptions required by ANOVA. If one or more of the assumptions are not met, the data will be transformed (log, square root) to improve normality and/or homogeneity. Post-ANOVA means will be separated using the Ryan-Einot-Gabriel-Welsch Multiple Range Test (SAS 1996). All tests of significance will be made at a=0.05.

Specific Work Products

The project will produce a final technical report containing information on (1) P concentrations of the most abundant soils (six soil types) of the SRV, (2) measurements of soil erosion for the major land cover types (crop, pasture, sage brush, forest), (3) sediment erosion and P transport from stream bank soils, (4) sediment and P sinks in the stream bed of the Sprague River and its tributaries and (5) sources and release of BAP from different soil types and land uses and though in-stream processing. We also plan to publish the data in a peer reviewed journal such as *Soil Science Society of America* or *Water, Air and Soil Pollution*.

Project Duration

July 1 2006 – June 30 2008.

The data will be collected and analyzed over two years, July 1 2006 – June 30 2008 with completion of the final technical report at the end of year 2. However, the first year of sampling will be designed as a stand-alone project should no further funding be available for year 2. To achieve this, we will collect and analyze samples from the dominant soil type for each land use (e.g. Klamath, Choptie & Maset) in year 1. We also will perform all lab incubations, including all soil types, in year 1. If funded in year 2, we will sample and analyze the remaining three soil types, Ontko, Yainix and Woodcock, for BAP, organic P, total P, supplemental soils data and ^{137}Cs. A project timeline is shown in table 2.

Permits

No permits are required to conduct this sampling**.**

Table 2. Schedule of work to be performed and deliverables to be produced.

	2006		2007				2008	
	July	Oct.	Jan.	Apr.	July	Oct.	Jan.	Apr.
Collect soils (all land uses, one soil type)	-----							
Prepare and analyze soils for TP, organic P, BAP & supplemental measurements		-----	-----	-----				
Prepare and analyze soils for ^{137}Cs		-----	-----	-----	-----			
Lab incubations		-----	-----	-----	-----			
Analyze data / write annual report				-----	-----			
Collect soils (all land uses, 2nd soil type)					-----			
Analyze soils for TP, organic P, BAP & supplemental measurements						-----	-----	-----
Prepare and analyze soils for ^{137}Cs						-----	-----	-----
Analyze data / write final technical report & peer-reviewed MS							-----	-----

Landowner Participation

We will work with KBERO and other agencies in the basin (e.g., NRCS) to identify private landowners who are willing to allow sample collection on their property.

Data Handling and Storage

Data will be entered into spreadsheet (Excel) files and stored electronically. Hard copies of the data will be included as appendices in the final technical report which will be made available to interested parties. Electronic files of the data also will be delivered as part of the final technical report and, like the paper copy, will be made available upon request.

Cost-sharing

Indiana University (IU) will cost share one academic month of Professor Craft's salary each year ($8,200/yr), including fringe benefits (($3,071/yr) and indirect cost recovery ($5,805/yr). The total cost share over two years will be $34,151.

Project Location

This project will focus on the Sprague River watershed and its tributaries, which is believed to be a major source of phosphorus P to UKL. The proposed study design is equally applicable to other UKL tributaries. Addiitonal sediment samples will be collected from the lower Williamson River and UKL for laboratory incubations.

Other Partners/Cooperators

We will coordinate our sampling where possible with other ongoing efforts by USGS, ODEQ, KBERO, NRCS, the Tribes, and other stakeholders that are focused on water quality issues in the Sprague River Valley.

Significance of Results

Among other findings, the results of this study will provide resource management and regulatory agencies in the Upper Klamath Basin with the following information:

1) Levels of total and biologically available P in soils from different land uses (crop, pasture, sage brush, forest), soil types, and geomorphic positions (stream bank, dstream bed).
2) The potential contribution of soils under different land uses, soil types and geomorphic positions to river and lake sediment and P loads.
3) The relative importance of sediment-bound vs. dissolved P sources in the Sprague River watershed.
4) How sediment P concentrations and bioavailability are altered as sediments are transported down to the lake.

Information gained from this work will allow these agencies to better assess the contribution of anthropogenic vs. natural processes to watershed P loading and to focus limited restoration funds on controlling the major human sources of P to UKL.

Literature Cited

Bortelson, GC and MO Fretwell. 1993. A review of possible causes of nutrient enrichment and decline of endangered sucker populations in Upper Klamath lake, Oregon. U.S. Geological Survey, Portland, Oregon, USA. Water-Resources Investigations Report 93-4087.

Bradbury, JP, SM Coleman and RL Reynolds. 2004. The history of recent limnologic changes and human impact on Upper Klamath Lake, Oregon. *Journal of Paleolimnology* 31 151-164.

Cooper, JR, and JW Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. *Soil Science Society of America Journal* 51:1600-1604.

Eilers, J.M., J. Kann, J. Cornett, K. Moser, A. St. Amand, and C. Gubala. 2001. Recent Paleolimnology of Upper Klamath Lake, Oregon. Prepared by J.C. Headwaters, Inc., for U.S. Bureau of Reclamation, Klamath Basin Area Office, Klamath Falls, OR.

Gee, G.W., and D. Or. 2002. Particle-size analysis. p. 255-293. *In* J.H. Dane and G.C. Topp [eds.], Methods of soil analysis, part 4: physical methods. Soil Science Society of America. Madison, Wisconsin.

Graham, SA, CB Craft, P McCormick and A Aldous. 2005. Forms and accumulation of soil P in natural and recently restored peatlands - Upper Klamath Lake, Oregon. *Wetlands* 25:594-606.

Kuo, S. 1996. Phosphorus. Pp. 869-920. *In*, Methods of soil analysis part 3: Chemical methods. SSSA book series 5. American Society of Agronomy. Madison, Wisconsin.

ODEQ (Oregon Dept. of Environmental Quality). 2002. Upper Klamath Lake Drainage Total Maximum Daily Load (TMDL) and Water Quality Management Plan (WQMP). Portland, Oregon.

NRC (National Research Council). 2004. Endangered and threatened fishes in the Klamath River basin. National Academies Press, Washington, DC.

SAS (Statistical Analysis Systems). 1996. SAS User's Guide. Cary, N.C.

Sharpley, A.N., M.J. Hedley, E. Sibbesen, A. Hillbricht-Ilkowska, W.A. House, and L. Ryszkowski. 1995. Phosphorus transfer from terrestrial to aquatic ecosystems. p. 171-199. In Tiessen, H. (ed.) Phosphorus Cycling in Terrestrial and Aquatic Ecosystems. United Nations SCOPE, Scientific Advisory Committee on Phosphorus Cycling. J. Wiley and Sons, London.

Sommers, LE, and DW Nelson. 1972. Determination of total phosphorus in soils: A rapid perchloric acid procedure. *Soil Science Society of America Proceedings* 36:902-904.

Sonzogni, W.C., S.C. Chapra, D.E. Armstrong, and T.J. Logan. 1982. Bioavailability of phosphorus inputs to lakes. *Journal of Environmental Quality* 11:555-563.

USDA (United States Department of Agriculture). 1985. Soil survey of Klamath County, Oregon: Southern part. USDA Soil Conservation Service, Washington, D.C.

USFWS (United States Fish and Wildlife Service). 1998. Klamath Basin Ecosystem Restoration Strategy. Klamath Falls, Oregon.

Vance, E.D., P.C. Brooks, and D.S. Jenkinson. 1987. An extraction method for measuring soil microbial biomass C. *Soil Biology and Biochemistry* 19:703-707.

Biographical Sketch

Christopher B. Craft

Indiana University, School of Public and Environmental Affairs
1315 East 10th Street, Bloomington, IN 47405
(812) 855-5971 direct, (812) 855-7802 fax
e-mail: ccraft@indiana.edu; web: http://www.spea.indiana.edu/Craft/

Education

1980 B.A., Biology, University of North Carolina-Asheville.
1983 M.S., Ecology, University of Tennessee. Knoxville.
1987 Ph.D., Soil Science, North Carolina State University. Raleigh.

Professional Experience

1999-present, Associate Professor, Indiana University, Bloomington IN
1997-1999, Assistant Scientist, Joseph W. Jones Ecological Research Center, Newton GA
1996-1997, Assistant Professor, University of Louisville, Louisville KY
1989-1995, Research Assistant Professor, Duke University, Durham NC
1987-1989, Research Associate, North Carolina State University. Raleigh, NC.

Five Most Relevant Publications

Graham, S.A., C.B. Craft, P. McCormick and A. Aldous. 2005. Forms and accumulation of soil P in natural and recently restored peatlands - Upper Klamath Lake, Oregon. Wetlands 25:594-606.

Aldous, A., P. McCormick, C. Ferguson, S. Graham and C. Craft 2005. Hydrologic regime controls soil phosphorus fluxes in restoration and undisturbed wetlands. Restoration Ecology 13:341-347.

Craft, C.B. and C. Chiang. 2002. Forms and amounts of soil nitrogen and phosphorus across a longleaf pine – depressional wetland landscape. Soil Science Society of America Journal 66:1713-1721.

Markewitz, D., F. Sartori and C. Craft. 2002. Soil change and carbon storage in longleaf pine planted on marginal agricultural land. Ecological Applications 12:1276-1285.

Craft, C.B. and W.P. Casey. 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. Wetlands 20:323-332.

Synergistic Activities

Associate Editor, *Soil Science Society of America Journal.* 2004-present.

Chair, Division S-10 (Wetland Soils), Soil Science Society of America. 2003-2004.

Wetlands Expert Working Group: Developing Nutrient Criteria for Wetland Systems. US Environmental Protection Agency. Washington D.C. January 2000 - present.

Biographical Sketch

Paul V. McCormick

USGS Leetown Science Center
11649 Leetown Rd
Kearneysville, WV 25430
(304) 724-4478 direct, (304) 724-4465 fax
e-mail: pmccormick@usgs.gov

Education

1984 B.A., Biology, Virginia Tech
1986 M.S., Zoology, Virginia Tech
1989 Ph.D., Aquatic Ecology, University of Louisville.

Professional Experience

2002-present, Research Ecologist, USGS Leetown Science Center, Kearneysville WV
2001-2002, Senior Ecologist, DOI Everglades Program Team, Boynton Beach FL
2000-2001, Williamson River Delta Restoration Project Director, Klamath Falls OR
1993-2000, Senior & Supervising Environmental Scientist, South Florida Water Management District, West Palm Beach FL
1991-1993 Assistant Professor, Elizabethtown College, Elizabethtown PA
1989-1991 Research Associate and Assistant to the Director, University Center for Environmental & Hazardous Materials Studies, Virginia Tech, Blacksburg VA

Five Most Relevant Publications

McCormick, P.V., and A. Parker. 2005. Identifying Sediment Phosphorus Release Pathways in Upper Klamath Lake to Support Lake Management Strategies. Draft Final Report to the Klamath Basin Ecosystem Restoration Office, Klamath Falls, OR.

Aldous, A., P. McCormick, C. Ferguson, S. Graham and C. Craft 2005. Hydrologic regime controls soil phosphorus fluxes in restoration and undisturbed wetlands. Restoration Ecology 13:341-347.

Graham, S.A., C.B. Craft, P. McCormick and A. Aldous. 2005. Forms and accumulation of soil P in natural and recently restored peatlands - Upper Klamath Lake, Oregon. Wetlands 25:594-606.

McCormick, P.V., and J.A. Laing. 2002. Effects of increased phosphorus loading on dissolved oxygen in a subtropical wetland, the Florida Everglades. Wetlands Ecology and Management

McCormick, P.V., S. Newman, S.L. Miao, K.R. Reddy, D.E. Gawlik, and T.D. Fontaine III, and D.J. Marley. 2001. Effects of anthropogenic phosphorus inputs on the Everglades. Pages 83-126 in *The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook* (J.W. Porter and K.G. Porter, eds.). CRC/Lewis Publishers, Boca Raton, Florida.

Synergistic Activities

Shenandoah River Fish Kill and Water Quality Multi-Agency Task Force. 2005-present.
Wetlands Expert Working Group: Developing Nutrient Criteria for Wetland Systems. US Environmental Protection Agency. 1999 - present.

APPENDIX 4 ESTIMATED BUDGET WORKSHEET

YEAR One	***TOTAL BUDGET***	***USFWS Funds Requested***	***Other Federal Funds***	***Non-Federal Cost Share***	
				Cash	***In-Kind***
1)Personnel:					
Chris Craft 2 wks summer salary	$13,110	$ 4,910			$ 8,200
1 PhD Student	$13,007	$13,007			
Fringe benefits:					
Craft 19.75%	$ 4,041	$ 970			$3,071
PhD student –health insurance	$ 1,057	$ 1,057			
Subtotal Personnel:	$31,215	$19,944			$11,271
2) Subcontractors: # Hours @ Hourly Rate Subtotal Subcontractors	$	$	$	$	$
3) Materials and Supplies: #Units@ Cost/Unit Subtotal Materials and Supplies	$ 9,500	$ 9,500	$	$	$
4) Operating Expenses: Subtotal Operating Expenses					
Out of State Travel	$ 7,660	$ 7,660			
Fee remission	$ 5,535	$ 5,535			
Shipping	$ 1,500	$ 1,500			
SUBTOTAL DIRECT COSTS	$55,410	$ 44,139	$	$	$11,271
5) Administrative Overhead Expenses:	$11,596	$ 5,791	$	$	$ 5,805
TOTAL PROJECT BUDGET	$67,006	$49,930	$	$	$17,076
Administrative Overhead	% 15	% 15			
%	%				

YEAR Two	***TOTAL BUDGET***	***USFWS Funds Requested***	***Other Federal Funds***	***Non-Federal Cost Share***	
				Cash	***In-Kind***
1)Personnel:					
Chris Craft 2 wks summer salary	$13,110	$ 4,910			$ 8,200
1 PhD Student	$10,775	$10,775			
Fringe benefits:					
Craft 19.75%	$ 4,041	$ 970			$3,071
PhD student –health insurance	$ 1,110	$ 1,110			
Subtotal Personnel:	$29,036	$17,765			$11,271
2) Subcontractors: # Hours @ Hourly Rate Subtotal Subcontractors	$	$	$	$	$
3) Materials and Supplies: #Units@ Cost/Unit Subtotal Materials and Supplies	$ 3,500	$ 3,500	$	$	$
4) Operating Expenses: Subtotal Operating Expenses					
Out of State Travel	$ 3,000	$ 3,000			
Fee remission	$ 5,923	$ 5,923			
Shipping	$ 1,000	$ 1,000			
SUBTOTAL DIRECT COSTS	$42,459	$ 31,188	$	$	$11,271
5) Administrative Overhead Expenses:	$ 9,595	$ 3,790	$	$	$ 5,805
TOTAL PROJECT BUDGET	$ 52,054	$34,978	$	$	$17,076
Administrative Overhead	% 15	% 15			

Effects of Sea level Rise and Climate Variability on Ecosystem Services of Tidal Marshes, South Atlantic Coast

Christopher Craft
School of Public and Environmental Affairs
Indiana University
Bloomington IN 47405

Samantha Joye
Marine Sciences Program
University of Georgia
Athens GA 30602-3636

Steve Pennings
Department of Biology and Biochemistry
University of Houston
Houston TX 77204-5001

Dick Park
Eco Modeling
5522 Alakoko Place
Diamond Head MS 39525

Jeff Ehman
Pangaea Information Technologies Ltd.
SPEA Building, 1315 East 10th Street
Bloomington IN 47405

Research Category and Sorting Code: 2004-STAR-J1
Title: Effects of Sea level Rise and Climate Variability on Ecosystem Services of Tidal Marshes, South Atlantic Coast
Investigators: C Craft (PI, ccraft@indiana.edu), S Joye (Co-PI, mjoye@uga.edu), S Pennings (Co-PI, spennings@uh.edu), D Park (Co-PI, dickpark@cableone.net), J Ehman (Co-PI, jaehman@indiana.edu)
Institution: Indiana Univ., Bloomington IN; Univ. of Georgia, Athens GA; Univ. of Houston, Houston, TX, Eco Modeling, Diamond Head, MS; Pangaea Info. Tech. Ltd., Bloomington IN.
Project Period: January 1 2005-December 31 2007.
Project Cost: $749,974
Project Summary: A conceptual model is proposed that describes how ecosystem services of tidal marshes vary along the salinity gradient and how climate change will alter the delivery of ecosystem services. Accelerated sea level rise is predicted to reduce the area of tidal marsh via submergence and conversion of tidal freshwater marsh to brackish & salt marsh. The result will be a reduction in ecosystem services of salt and brackish marshes along with an almost complete loss of services provided by tidal freshwater marshes. Predicted greater inter-annual variability of climate will lead to greater frequency of drought that reduces delivery of ecosystem services and freshwater pulsing which we predict will enhance delivery of ecosystem services.

We will test the effects of rising sea level and greater inter-annual variability of climate on alteration of area and ecosystem services of tidal marshes in three estuaries (Altamaha, Satilla and Savannah Rivers, GA). Ecosystem services related to disturbance (shoreline protection) and gas regulation (CO_2 & CH_4 flux), soil formation (C sequestration), nutrient regulation (N, P retention), waste treatment (sediment deposition, denitrification), refugium and food (macrophytes & marsh nekton) will be measured in replicate salt, brackish and tidal freshwater marshes of each watershed. GIS in conjunction with the SLAMM model will be used to predict changes in marsh area resulting from submergence and habitat conversion. Overlay of ecosystem-level measurements will be used to predict how cumulative delivery of ecosystem services in each estuary will be altered in response to incremental (10 cm) increases in sea level. SLAMM also will be used to predict changes in shoreline protection potential of tidal marshes, commercial shrimp yields and the effects of dikes on delivery of ecosystem services. The results of the model will be scaled to the South Atlantic Coast (GA, SC) region. The effects of climate variability will be evaluated by analysis of climate (rainfall, temperature, salinity, freshwater discharge, average tide level) and ecosystem services data collected since 2000 from permanent plots of ten marshes of the GCE LTER study domain. Climate data for the region, including temperature, precipitation and river discharge extend back several decades and have been compiled by the GCE LTER program, allowing current patterns to be placed in historical context. Marsh vegetation, epifauna, accretion and sediment deposition data are collected biannually since 2000, through drought and average rainfall years.

This work will (1) provide a basis to understand how ecosystem services vary among salt-, brackish- and tidal fresh-water marshes, (2) determine how sea level rise will alter marsh area and delivery of ecosystem services, (3) evaluate the effect of diking on delivery of ecosystem services and (4) elucidate how climate variability will affect temporal patterns of macrophytes, epifauna, sediment deposition and marsh accretion. A GIS-based model describing the effects of rising sea level on tidal marsh ecosystem services of river dominated estuaries will be produced that can be applied to comparable estuaries of the U.S.
Keywords: Climate change, Wetlands, Indicators, Regionalization, GA, SC, Region IV

RESEARCH PLAN

Objectives

Ecosystem Services of Tidal marshes

Tidal marshes consist of salt marshes, brackish-water marshes and tidal freshwater marshes. They are arrayed along a gradient of salinity and vary in terms of the ecosystem services they provide. Ecosystem services of tidal marshes consist of functions associated with (1) regulation, (2) habitat, (3) production and (4) information (de Groot et al. 2002) and specific examples include shoreline protection & waste treatment, habitat for wild plants, biological productivity and recreation (Richardson 1994, Daily et al. 1997). It has been estimated that tidal marshes provide nearly $10,000/ha/yr USD of ecosystem services to society (Costanza et al. 1997). Salt marshes, dominated by smooth cordgrass (*Spartina alterniflora* Loisel), are known for their high levels of primary and secondary production (Daiber 1982, Wiegert and Freeman 1990), role as nursery and feeding grounds for estuarine organisms (Kneib 1997), ability to trap sediment, nutrients and pollutants (Nixon 1980, DeLaune et al. 1981, Khan and Brush 1994), sequester carbon (Conner et al. 2001) and export organic C to estuarine foodwebs (Peterson et al. 1985, Childers 1994). Much less is known regarding ecosystem services of brackish and tidal freshwater marshes.

In a review article, Odum (1988) speculated about ecosystem services of salt marshes versus tidal freshwater marshes. He hypothesized that primary production (NPP) is lower in salt marshes because of stress associated with salinity and sulfides, both of which will increase in response to rising sea level. This hypothesis was supported by measurements of NPP in tidal freshwater marshes that were comparable to or greater than in saline tidal wetlands (Whigham et al. 1978, Doumlele 1981, Perry and Atkinson 1997). Reduced levels of stressors also result in greater plant diversity in tidal freshwater marshes (Simpson et al. 1983, Odum 1988). Diversity of consumers, invertebrates and fish, is thought to be lower in tidal freshwater marshes as compared to salt and brackish marshes (Odum 1988) but it is not known how secondary production varies among tidal marshes. Odum hypothesized that organic matter accumulation is higher in tidal freshwater versus salt marshes as a result of greater NPP and reduced decomposition which is supported by published studies (Bowden 1984, Craft et al. 1988). Finally, he hypothesized that sediment deposition is greater in tidal freshwater marshes than salt marshes because of the proximity to riverine sediment inputs and flocculation of suspended sediment by saltwater intrusion during periods of low flow. Paludan and Morris (1999), however, observed that sediment deposition was highest in brackish marshes near the turbidity maximum in the estuary.

Preliminary studies along the Altamaha River (Georgia), as part of the Georgia Coastal Ecosystems Long Term Ecological Research (GCE LTER) studies, support some of Odum's hypotheses regarding ecosystem services of salt versus tidal freshwater marshes. For example, macrophyte species richness is greater in the tidal freshwater marsh than in the brackish marsh or salt marsh (S. Penning, unpublished data). Sediment deposition and soil organic C and N accumulation is greater in the brackish and tidal freshwater marsh as compared to salt marshes in the region (C.B. Craft, unpublished data). The tidal freshwater and brackish marsh trapped two times more sediment (1000 $g/m^2/yr$) and sequestered four times more C (130 $g/m^2/yr$) and N (8 $g/m^2/yr$) than the salt marsh (620 g sediment/m^2/yr, 30 $g/m^2/yr$, 2 g N/m^2/yr). These results agree with other studies that show greater organic matter and N accumulation in tidal freshwater and brackish marshes than in salt marshes (Hatton et al. 1982, Craft et al. 1993, Merrill and

Cornwell 2000). Denitrification, another ecosystem service related to waste treatment, is greater in freshwater than saline environments (Seitzinger 1988) and also is greater in intertidal versus subtidal sediment (Merrill and Cornwell 2000). Denitrification is strongly coupled to nitrification of ammonium (Jenkins and Kemp 1984) and, in saline environments, sulfide inhibits nitrification (Joye and Hollibaugh 1995). When nitrate is not limiting, denitrification is regulated by availability of labile organic carbon (Groffman 1994), that has been shown to vary among salt, brackish and tidal freshwater marshes. Methanogenesis also is greater in freshwater than saline wetlands (Bartlett et al. 1987, Capone and Kiene 1988, Oremland 1988).

Findings of greater biodiversity and carbon sequestration in tidal freshwater marshes of the Altamaha River and other estuaries agree with predictions made by Odum regarding ecosystem services of salt versus tidal freshwater marshes. However, some findings from the Altamaha River estuary suggest that Odum's predictions regarding ecosystem functions of tidal freshwater and salt marshes may not be correct. For example, there was no difference in aboveground NPP among salt (*S. alterniflora*, 2840 $g/m^2/yr$), brackish (*S. cynosuroides*, 3080 $g/m^2/yr$) and tidal freshwater (giant cutgrass-*Zizaniopsis milacea*, 2490 $g/m^2/yr$) marshes (Schubauer and Hopkinson 1984, Hopkinson 1992), contradicting the hypothesis of greater NPP in tidal freshwater versus salt marshes. Almost no comparisons of ecosystem services of salt, brackish and tidal freshwater marshes have been made in single estuarine systems using standard methods. Thus, although some of Odum's hypotheses can be examined using literature data, this typically requires comparing studies done at different times in different estuaries by different investigators using different methods. Consequently, our ability to rigorously evaluate Odum's hypotheses remains quite limited.

Below, we develop a conceptual model describing how ecosystem services vary along the estuarine gradient from tidal freshwater marshes to salt marshes (Figure 1). We hypothesize that tidal freshwater marshes provide higher level of regulation functions, including gas regulation, soil formation, nutrient regulation and waste treatment than salt marshes. Tidal freshwater marshes also provide higher levels of habitat and production functions of macrophytes. We hypothesize that salt marshes provide a higher level of disturbance regulation through their seaward extent that gives them greater ability to buffer waves and greater utilization by marsh nekton.

Effects of Climate Change on Tidal Marsh Ecosystem Services

Tidal marshes exist at the interface between land and sea and, so, are uniquely suited to provide ecosystem services associated with disturbance regulation and waste treatment. These services are especially important to the 53% of the population of the United States that live in the coastal zone (NOAA 2000). Tidal marshes also are among the most susceptible ecosystems to climate change. Climate change models predict that, in the next 100 years, sea level will increase 30 cm to 85 cm (http://ipcc-ddc.cru.uea.ac.uk/asres/sres_home_climate.html). Rising sea level will have the greatest effect through submergence (Park et al. 1989a, Brinson et al. 1995, Moorhead and Brinson 1995) and loss of brackish and tidal freshwater marshes as salt marshes transgress landward (Park et al. 1991). Landscape models predict that the net effect of rising sea level on tidal marshes will be reduced area of marsh habitat and a landward shift in salt, brackish and tidal freshwater marsh habitat. Changes in the cover of different habitat types will consequently lead to changes in the quantity and types of ecosystem services provided by these wetlands.

Future climates are expected to differ from current ones not only in the mean values but also in variability. In particular, temperature and precipitation are expected to increase in vari-

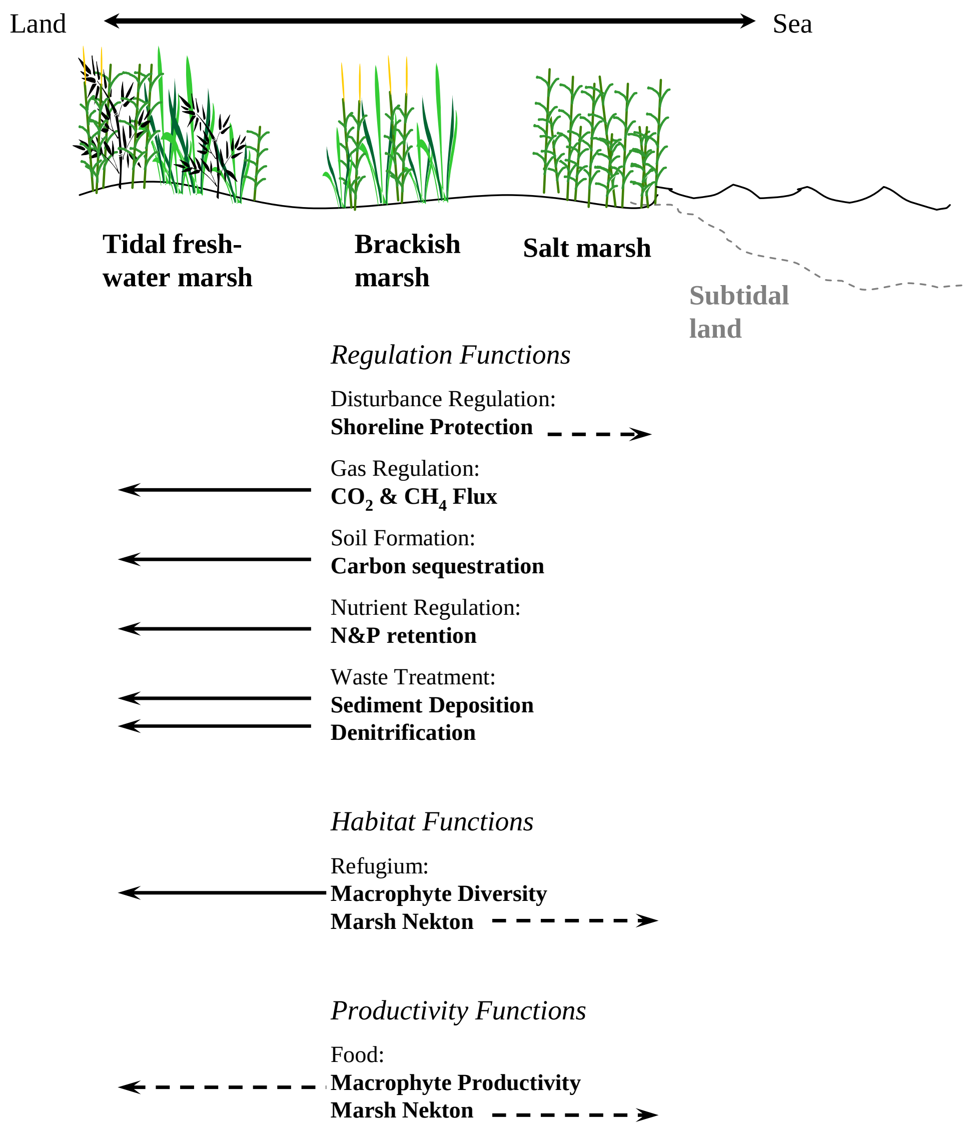


Figure 1. Conceptual diagram showing how ecosystem services vary among salt, brackish and tidal freshwater marshes and subtidal land. The relative importance of a particular ecosystem service (e.g. macrophyte diversity) increases in the direction of the arrow (e.g. tidal freshwater marsh). Dashed arrows indicate less certainty with respect to the direction and magnitude of the service (e.g. macrophyte productivity).

ability (Karl et al. 1995, Mahlman 1997). Although climatic variability may not affect ecosystem services to the same extent as sea level rise, it does have the potential to alter the magnitude and types of ecosystem services provided by tidal marshes. For example, greater inter-annual variability in climate may lead to greater frequency, severity and duration of drought. Drought alters wetland soil physicochemical properties by promoting dessication, hypersalinity, oxidation of pyritic materials, soil acidity and metal (Fe, Mn, Zn and others) toxicity all of which may stress marsh vegetation (Smith 1970, Linthurst and Seneca 1980, Morris 2001, McKee et al. 2004). The stress caused by extended drought may lead to dieback of marsh vegetation over large areas producing a phenomenon known as "brown" marsh (Stewart et al. 2001, http://www.brownmarsh.net). In Louisiana, the brown marsh phenomenon was linked the effects of the ongoing drought, low flow conditions of the Mississippi River and lower than average sea level that altered salinity, pH and bioavailability of metals. Although we are familiar with brown marsh phenomenon in Louisiana (Smith 1970, Michot and Wells 2001, Hester et al. 2004), Georgia (Flory and Alber 2002, appendix 2; GCRC 2004, McKee et al. 2004) and elsewhere (Linthurst and Seneca 1980, Carlson et al. 2001), the relationship between climate change and drought is beyond the scope of our study. We will focus instead on the more typical effects of annual variation in climate, which have important effects regardless of whether or not brown marsh develops.

Drought was implicated in marsh dieback along the Georgia and South Carolina coasts following an extended drought from 1998-2002 (Flory and Alber 2002, appendix 2; GCRC 2004). During the drought, temperature was 1.6^{o}F warmer than the 30-year average (Figure 2a) and rainfall was rainfall was 80% (41.91 inches) of the 30 year average (Figure 2b). Data collected from permanent plots at ten marsh locations of the GCE LTER study domain indicated that, even in areas not affected by visible dieback, drought depressed macrophyte aboveground production, stem height & density and number of flowering stems (Figure 2c) that rebounded the year after drought abated (2003). Greater productivity of *Spartina* in 2003 was attributed to a doubling of freshwater discharge from the Altamaha River that lowered salinity considerably as compared to 2002 (Figure 2c). A decline in landings of shrimp and oysters occurred during the drought (Figure 2d). Landings of both species increased in 2002 the drought began to abate.

In contrast to drought, greater frequency of above-average rainfall, river flooding and freshwater groundwater input reduce plant stress by lowering salinity and consequently favor expansion of brackish marsh species into salt marshes where *S. alterniflora* normally dominates (DeLaune et al. 2003, Steve Pennings, unpublished data). Such a shift in salinity would enhance some ecosystem services at the expense of others (see Figure 1). For example, macrophyte production was enhanced by periods of increased freshwater input relative to drought conditions (Figure 3c).

Management Decisions – Dike Maintenance and/or Removal

Along the Georgia and South Carolina coasts and elsewhere, marshes along river-dominated estuaries such as the Altamaha River were historically diked for rice production (Odum et al. 1984). The effects of diking include isolation from tidal inundation and freshwater input & lowering of salinity (Sinicrope et al. 1990). Diked marshes provide some level of ecosystem services such as gas (CO_2 & CH_4) regulation, soil formation and refugium & production functions although at lower levels than undiked marshes (Sturdevant et al. 2002, Stocks and Grassle 2003). Diked marshes trapped less sediment, retained less N and P and sequestered less carbon than undiked marshes (Sturdevant et al. 2002). Marsh food webs also are altered. Abundance of benthic macrofauna were reduced in impounded versus natural marshes (Stocks and Grassle 2003). Finfish utilization, especially for transient species, was less in impounded marshes (Rey et al. 1990). Water quality also was degraded by impoundment as dissolved oxygen was less and nutrient concentrations were greater in impounded versus natural marshes (Rey et al. 1991, Brockmeyer et al. 1997).

Management strategies to mitigate for wetland loss against rising sea level may include maintenance of existing dikes to protect freshwater marshes against salt water encroachment or restoring tidal inundation to enhance sediment deposition, promote vertical accretion and maintain elevation in the face of rising sea level. However, restoring tidal inundation to diked organic soil marshes may lead to increased sulfate reduction, decomposition of soil organic matter and

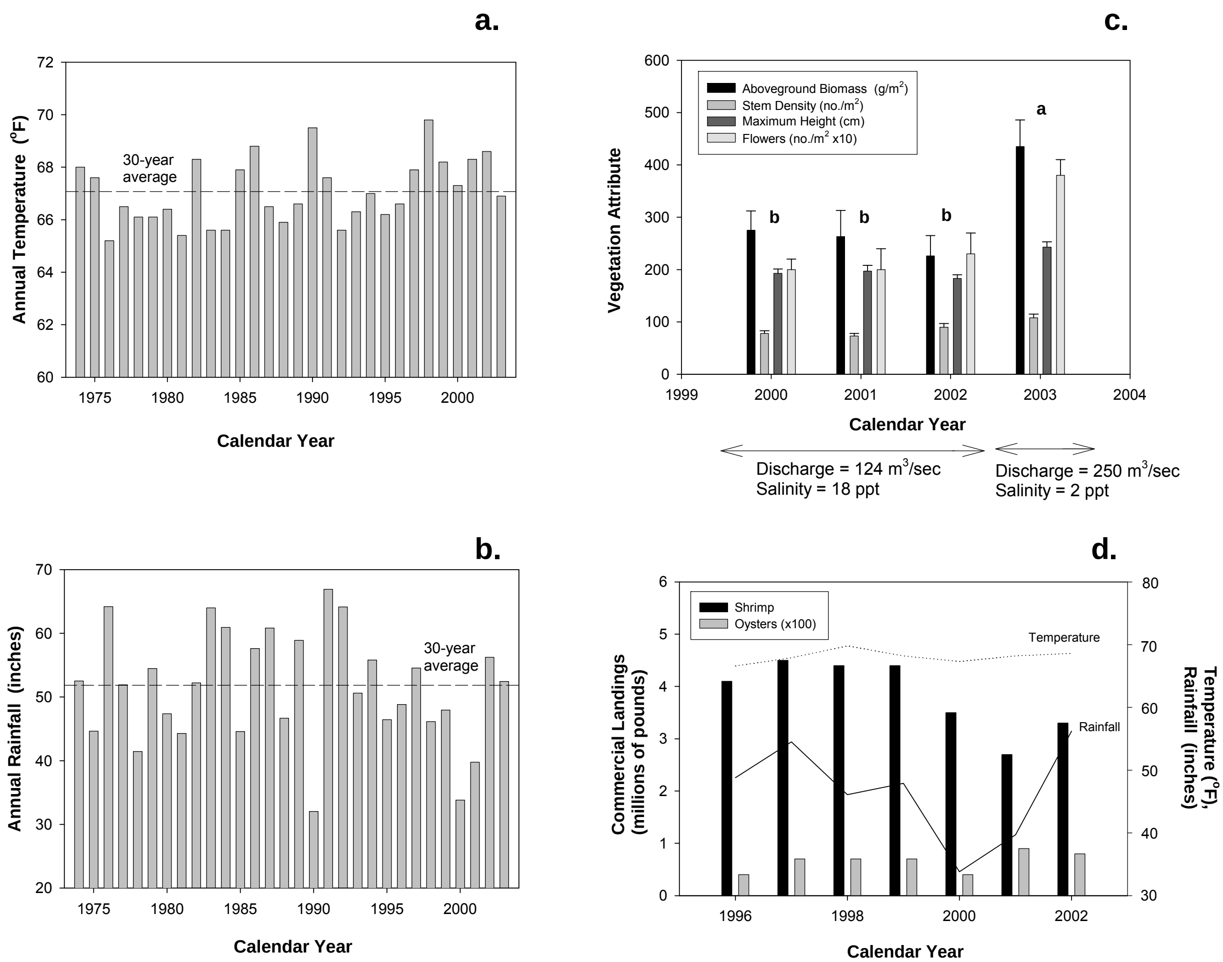


Figure 2. Mean annual (a) temperature and (b) rainfall (1974-2003) at Sapelo Island, Georgia). (c) Vegetation attributes of "tall" *S. alterniflora* at ten sites during drought (2000-2002) and average (2003) rainfall years. (d) Commercial landings of shrimp and oysters (1996-2002) along the Georgia coast.

subsidence (Portnoy 1999). As a result, dike removal may exacerbate submergence of marshes unless sediment deposition is sufficient to offset subsidence. Management actions remove or maintain dikes must weigh the benefits of protecting freshwater marshes against the cost associated with the reduction or loss of waste treatment and refugium functions that depend on connectivity to the estuary.

Hypotheses

Our primary hypothesis is that rising sea level will lead to a reduction in the area and distribution of salt, brackish and tidal freshwater marshes that alters the delivery of ecosystem services.

Hypothesis I: Rising sea level will result in inundation and loss of tidal marshes, especially tidal freshwater marshes with the concurrent loss and/or reduction of ecosystem services associated with regulation, habitat and production functions (Figure 3).

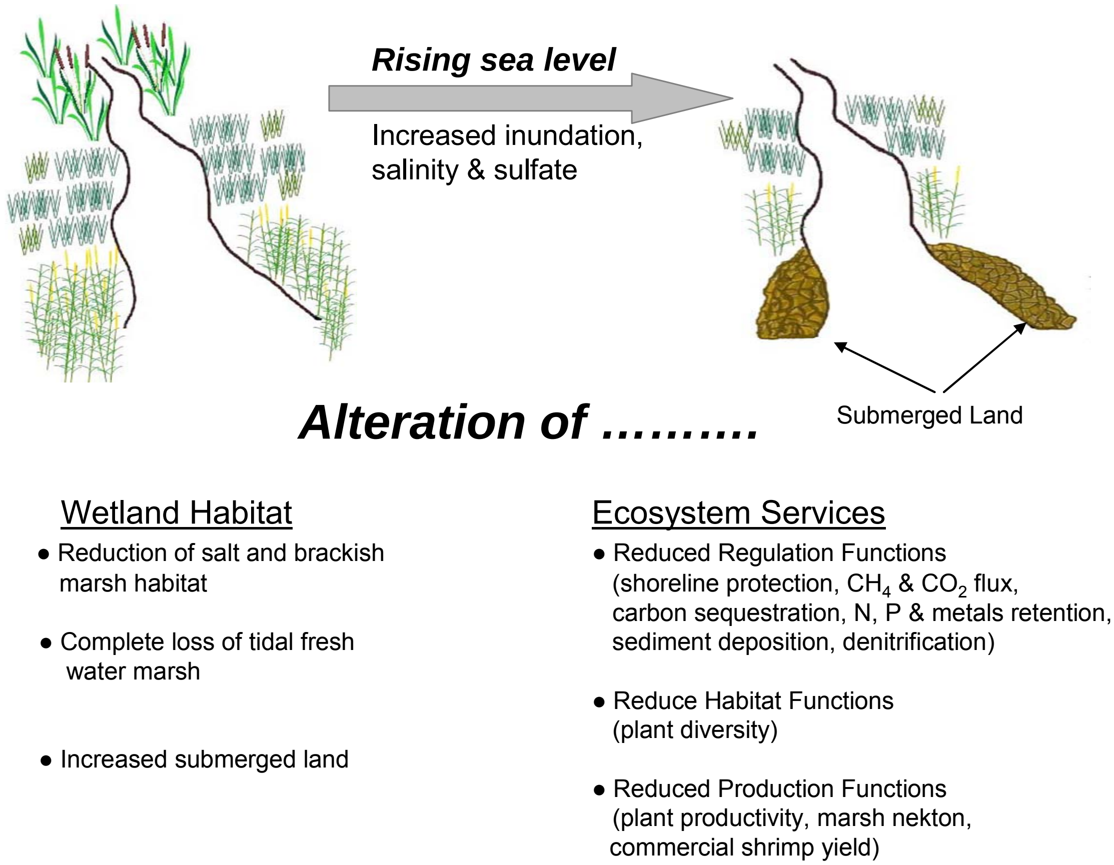


Figure 3. Hypothesized effects of rising sea level on area of tidal marsh habitat and delivery of ecosystem services.

Application of the SLAMM (Sea Level Affecting Marshes Model) to the southeastern U.S. coast predicts loss of tidal marshes, especially tidal freshwater marshes, in response to rising sea level (Park et al. 1989a, 1991). Based on our conceptual model, loss of tidal freshwater marshes is predicted to lead to a reduction in regulation functions (gas regulation - CO_2 and CH_4 flux, soil formation - carbon sequestration, nutrient regulation - N and P storage, waste treatment - sediment deposition, denitrification), habitat functions (plant diversity) and food functions (plant & productivity) (Figure 3). Ecosystem services associated with salt marshes (disturbance regulation – shoreline protection) and food functions (marsh nekton & shrimp yields) may be enhanced in these areas as salt marshes transgress into areas formerly dominated by tidal freshwater marsh vegetation but, overall, there will be a net loss of tidal marsh ecosystem services in response to rising sea level. We predict a near complete loss of ecosystem services associated with tidal freshwater marshes, reduced area of salt and brackish marsh habitat and a reduction in services provided by these tidal marshes.

Hypothesis II: Diking protects and maintains freshwater marshes against rising sea level. However, when marshes are diked, ecosystem services associated with connectivity (waste treatment, refugium) are lost.

Ecosystem services related to waste treatment (sediment deposition, N&P retention, denitrification) require connectivity to adjacent estuaries and water bodies in order to provide the purification functions. Similarly, utilization of tidal marshes by nekton is lost when marshes are diked. Functions associated with plant production and diversity are maintained although, perhaps, at lower levels than undiked marshes. The net effect of diking is the protection of freshwater marsh habitat but with the loss of essential ecosystem services associated with waste treatment and refugium.

Hypothesis IIIa: Greater inter-annual variability of climate leads to greater frequency of drought and reduction in ecosystem services in drought years (Table 1)

Hypothesis IIIb: Greater frequency of above average rainfall enhances ecosystem services such as macrophyte productivity & diversity that respond to increased freshwater inputs in wet years (Table 1).

Drought-induced dieback of brackish and salt marshes leads to loss of ecosystem services associated with vegetation, productivity and diversity (Table 1). Services associated with waste treatment (sediment deposition) also are reduced by loss of vegetation whose stems dissipate wave energy (Knutson 1988). We also hypothesize that marsh nekton (i.e. epifauna) & productivity also will be reduced in response to drought conditions.

We hypothesize that above-average rainfall and river flooding enhances ecosystem services by increasing the delivery of freshwater and ameliorating salinity stress. Macrophytes, including *S. alterniflora*, respond to freshwater by enhancing NPP (Phleger 1971, Linthurst 1980, and Seneca 1981). In salt marshes of the Altamaha River, several years of above average rainfall and freshwater discharge led to increased abundance of oligohaline species such as *Scirpus robustus* in marshes previously dominated by *S. alterniflora* (S. Pennings, unpublished data). We also hypothesize that increased macrophyte production enhances sediment deposition by increasing stem density & diameter that reduces wave height & energy (Knutson 1988).

Approach

I. Effects of Rising Sea Level

We will use the space-for-time approach to predict which ecosystem services are altered or lost when sea level rises, tidal marsh habitat is submerged and transgression of salt marshes

Table 1. Hypothesized effects of climate variability (drought, freshwater pulsing) on ecosystem services of tidal marshes.

Drought:	Reduced…	Macrophyte productivity & diversity, marsh nekton & shrimp, sediment deposition, marsh accretion
Freshwater:	Enhanced…	Macrophyte productivity & diversity, marsh nekton & shrimp, sediment deposition, marsh accretion

landward displaces brackish and tidal freshwater marsh habitat. Rising sea level is predicted to reduce the area of tidal marshes, especially tidal freshwater marshes (Park et al. 1991). By comparing ecosystem services of salt, brackish and tidal freshwater marshes, then using SLAMM to model incremental (10 cm) increases in sea level rise, rates of submergence and habitat conversion, we can make predictions regarding which services will be lost or reduced as a result of sea level rise.

We will quantify ecosystem services of salt, brackish and tidal freshwater marshes of three estuaries of the south Atlantic coast, the Altamaha, Satilla and Savannah Rivers (Georgia) (Figure 4). The south Atlantic coast contains about 800,000 acres of tidal marshes (Hefner et al. 1994), 16% of our nation's estuarine wetlands (4,780,000 acres, Dahl and Johnson 1991). Only Louisiana (1,900,000 acres) and Florida (1,400,000 acres including mangroves) contain more estuarine wetland habitat (Hefner et al. 1994). The Altamaha and Savannah Rivers are among the largest estuarine systems along the South Atlantic coast and both estuaries, as well as the Satilla River, contain extensive salt marsh, brackish marsh and tidal freshwater marsh habitat, including at least 1,000 acres of tidal freshwater marsh in each estuary (Odum et al. 1984).

Within each estuary, two salt-, two brackish- and two tidal fresh-water marshes each will be selected. Marshes will be chosen based on dominant vegetation, which correlates strongly with salinity; *S. alterniflora* in salt marshes (20-35 ppt), *Juncus roemerianus* in brackish marshes (5-20 ppt) and *Zizaniopsis mileacea* in tidal freshwater marshes (<0.5 ppt). In each marsh, we will measure a suite of ecosystem services related to regulation, habitat and production functions (Figure 1). Sampling will be stratified by marsh zone (creekbank versus platform).

Ecosystem services associated with regulation functions will be evaluated by measurements of gas regulation (CO_2 and CH_4 flux), soil formation (C sequestration), nutrient regulation (N&P retention) and waste treatment (sediment deposition, denitrification). Incubations of sediment (without plants) will be conducted under submerged and exposed conditions to quantify benthic fluxes of CO_2 and CH_4 (Joye et al. 1996, Chanton and Whiting 1995, Edwards and Riggs 2003). Fluxes under exposed (low tide) conditions will be measured in the field using standard flux techniques (Morris and Whiting 1986, Chanton and Whiting 1995, Edwards and Riggs 2003). Sub-samples of chamber headspace will be collected at several time points and analyzed to determine methane and carbon dioxide concentrations using a trace gas analyzer (infrared detection; Edwards and Riggs 2003). Fluxes will be determined from the changes in concentration over time.

Fluxes under submerged conditions will be evaluated using cores collected in the field and incubated in the lab under at in situ temperatures, with continuous gentle stirring of the headspace (Joye et al. 1996). The water overlying the cores will be collected from the sampling sites and it will be sterile (0.2 µm) filtered prior to addition to the cores. Gas concentrations will be determined following headspace extraction of liquid samples.

The potential factors, e.g. salinity, sulfate concentrations, organic carbon concentrations, nutrients, regulating methane and carbon dioxide production in sediments will be evaluated in laboratory experiments carried out under anaerobic conditions (in an anaerobic (COY) chamber). Sediment slurries will be prepared and amended with substrates (organic carbon, sulfate), nutrients (N, P) or inhibitors (salts, sulfide) and the changes in accumulation of CH_4 and CO_2 will be monitored over time.

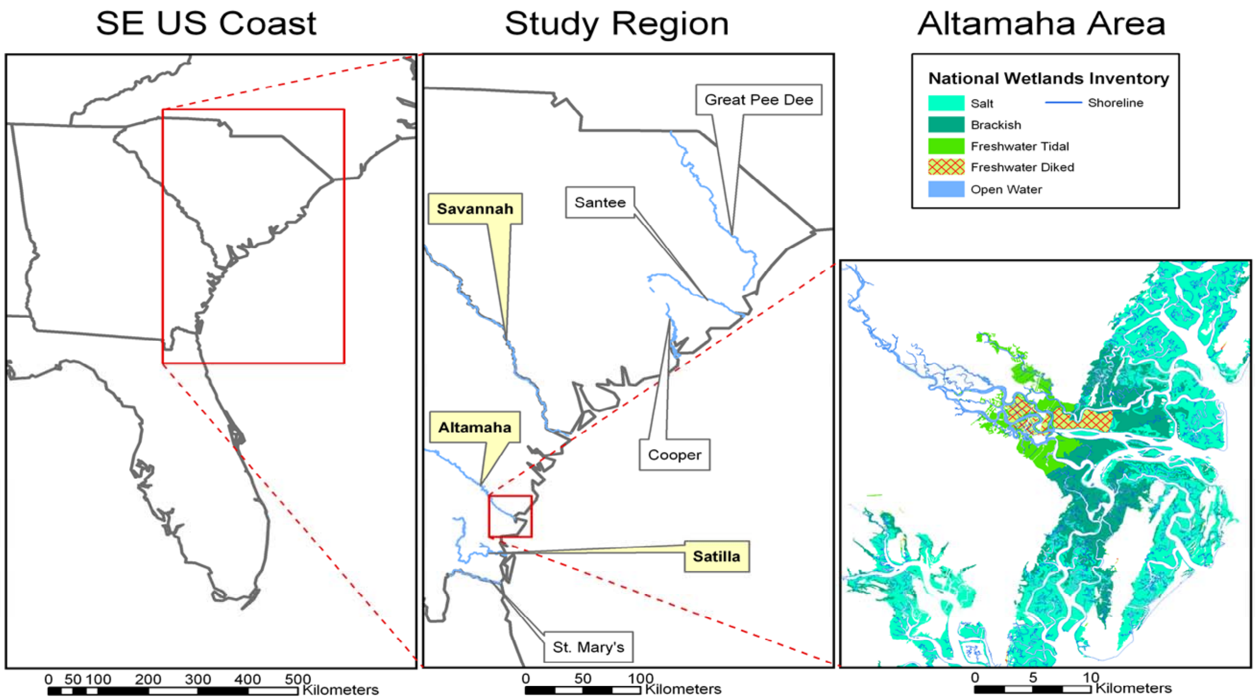


Figure 4. Study region, including the southeast (GA, SC) coast, field sampling areas (Altamaha, Satilla, Savannah Rivers) and GIS map of tidal wetlands of the Altamaha River Estuary (GA).

Carbon sequestration, sediment deposition and retention of N & P in soil will be measured by collecting two soil cores (8.5 cm diameter by 50 cm deep) from each marsh (n=36 cores) in year 1. Cores will be sectioned into 2 cm increments and each increment will be analyzed for ^{137}Cs, ^{210}Pb (to determine marsh accretion), bulk density, organic C and nutrients (N, P). Forty and 100 year rates of sediment deposition, C sequestration and nutrient accumulation will be calculated using accretion rate, bulk density and C, N & P concentrations. Sediment deposition also will be measured by installing three 0.25 m^2 feldspar marker layers in each marsh (Cahoon 1994) in year 1. Small diameter soil cores (n=2 per plot) will be collected six and 12 months later to determine rates of sediment accretion and accumulation.

Denitrification will be measured in core incubations (same as above) by quantifying changes in dinitrogen (N_2) concentrations over time using membrane inlet mass spectrometry (Kana et al. 1998). Replicate (n=3 to 4) cores from each habitat will be incubated for these experiments. We will conduct experiments at both in situ nitrate concentrations and with added nitrate. To evaluate the factors controlling denitrification rates, we will conduct separate slurry experiments where the concentration of nitrate, labile organic carbon and sulfide (a potential inhibitor of denitrification, Joye 2002) are varied (Joye and Paerl 1994). These experiments will be well replicated (n=3 to 4 per treatment) at each of the study sites.

Primary production of macrophytes will be measured by end of season sampling of aboveground biomass from 0.25 m^2 quadrats (n=10 per zone per marsh) in year 2. Aboveground biomass, stem height, density & diameter will be measured in each quadrat. Stem density and diameter data will be combined with GIS-based measurements of marsh width to determine shoreline protection potential through reduction in wave energy by emergent vegetation (see *IV.*

*Scaling Up: From Ecosystem-level Measurements to Regional Impacts)***.** Net primary production will be calculated from turnover rates published for *S. alterniflora, S. cynosuroides*, (Schubauer and Hopkinson 1984), *Juncus roemerianus* (Hsieh 1996) and *Zizaniopsis milaceae* (Birch and Cooley 1982, Hopkinson 1992). Macrophyte species richness will be measured in quadrats and also by species counts in a larger area, $100m^2$. Stocks of N and P present in aboveground plant biomass will be determined by elemental analysis of individual stems. Stocks will be scaled to the plot level using data on standing biomass per m^2. The same approach for scaling other ecosystem services from field plot to estuary and regional levels will be used.

Productivity and diversity of marsh nekton will be assessed by measurements of numbers and biomass of young crustaceans and fish that utilize the marsh surface using pit traps. Kneib (1997) compared small diameter (10 cm) pit traps for sampling small fish with actual densities in intertidal marsh enclosures and found that traps captured 34-49% of larvae and 61-72% of juveniles. There was a strong relationship (r^2=0.82-0.96) between numbers recovered and actual densities, indicating that pit traps provide a reliable index of small nekton use of the marsh. Ten trays, each 30x20 wide and 5 cm deep will be placed on the surface of each marsh at low tide. Trays will be depressed into the soil so that they are flush with the soil surface. Trays will be retrieved 24 hours later. Static water level recorders will be placed in each marsh at the same time to determine the depth of inundation during the 24 hour period. Organisms trapped in the trays will be separated (shrimp & other crustaceans, gastropods, bivalves, fish, insects) weighed wet, then preserved in ethyl alcohol. Organisms will be classified taxonomically (Eddy 1969, Lee et al. 1980, Williams 1984) to determine species richness then dried and reweighed to determine dry weight biomass. Sampling will be performed two times a year over a two year period.

A factorial ANOVA based on watershed (Altamaha, Savannah, Satilla Rivers), habitat type (tidal freshwater-, brackish-, salt-marsh, subtidal), time of sampling and location (streamside, marsh plain) will be used to test our hypotheses concerning ecosystem services of tidal marsh and subtidal habitat (SAS 1996). Our null hypothesis is that ecosystem services do not vary among tidal freshwater marsh, brackish marsh and salt marsh and subtidal sediments. Where appropriate, data will be transformed to meet the assumptions of the ANOVA (Sokal and Rohlf 1995). Typically, proportional data will be arcsine (square root) transformed, and in cases where the variance increases with the mean, numerical data will be log transformed, in order to improve normality and homogeneity of variance. Means will be separated using *a posteriori* means comparison tests such as the Ryan-Einot-Gabriel-Welsch (REGW) multiple range test (SAS 1996).

II. Effects of Dikes on Ecosystem Services

We will compare ecosystem services of diked freshwater marshes with tidal freshwater marshes to evaluate what ecosystem services are compromised when marshes are diked. Two diked marshes in each watershed will be sampled. Selected ecosystem services related to regulation (CO_2 & CH_4 flux, C sequestration, N&P retention, sediment deposition, denitrification), refugium and production (of macrophytes) functions shown in Figure 1will be measured using the methods described previously.

Analysis of variance will be used to test the effects of diking on delivery of ecosystem services. Data will be transformed as necessary and means will be separated using REGW multiple range test. These data will be collected in year 2 of the study. The data will be used to compare changes in the delivery of ecosystem services when marshes are diked versus undiked under different scenarios of sea level rise using the SLAMM model.

III. Effects of Climate Variability

Data collected from permanent plots of the GCE LTER study domain since 2000 will allow us to evaluate the effects of climate variability on selected ecosystem services of tidal marshes. As part of the GCE LTER study, biannual measurements of salinity, macrophyte biomass, stem height, density & species richness, marsh epifauna, marsh accretion and sediment deposition have been collected at ten marshes (seven salt marshes, two brackish marshes and one tidal freshwater marsh). The first two years (2000-2001) of the six-year study were marked by extreme drought followed by a return to "average" levels of precipitation beginning in 2002 (Figure 2b). Mean precipitation for 2000-2001 was 70% (36.8 inches) of the 30-year average and mean temperature was 0.7^{o}F warmer (67.8^{o}F) than the 30-year average. Estuarine salinity levels in the Altamaha River estuary also were much higher in 2001 (18 ppt at the mouth of the Altamaha River) as compared to 2002 (2 ppt) (GCE LTER, unpublished data). Productivity of creekbank *S. alterniflora*, as determined by aboveground biomass, stem height and stem density, was depressed in drought years (2000-2002) as compared to 2003, an average rainfall year (Figure 2c). Likewise, a decline in commercial landings of shrimp and oysters was noted during the drought. Landings rebounded somewhat in 2002 as the drought abated (Figure 2d).

We will use GCE LTER data to compare changes in ecosystem services in response to variability in rainfall and river flooding over several years. The GCE LTER study will continue to collect these data for at least two more years, and probably longer, assuming a successful proposal renewal, enabling us to construct a multi-year (6-8 years) dataset containing climate data and data describing selected ecosystem services. Aboveground biomass is non-destructively estimated in permanent plots (n=8 creekbank and n=8 platform) at each of ten marshes using allometric relationship based on stem density and height (Steve Pennings unpublished data). Marsh epifauna, including gastropods, bivalves and fiddler crabs (*Uca*) are monitored in permanent plots. Sediment deposition and marsh accretion are determined using feldspar marker layers (Cahoon 1994) and sedimentation-erosion tables (SET's) (Boumans and Day 1993), respectively.

Because of limited LTER funds, however, epifauna samples collected from the ten marshes are not sorted or identified. Funding from this EPA RFA will be used to support (1) sorting and taxonomic identification of marsh epifauna, (2) synthesis of existing and new monitoring data from the permanent plots and (3) statistical analysis of the data to elucidate relationships between annual rainfall, temperature, salinity and other environmental variables. Correlation, regression and multivariate statistical analysis will be used to explore relationships between climate variability (annual precipitation, temperature, salinity, sea level anomalies (Morris et al. 1990) and selected ecosystem services.

IV. Scaling from Ecosystem-level Measurements to Regional Impacts

Changes in marsh submergence and wetland habitat conversion in response to different scenarios of sea level rise will be modeled using the Sea Level Affecting Marshes Model (SLAMM) through a subcontract to Dick Park, who developed the model. Our estimates of sea level rise are taken from climate change models that are reported in the IPCC Special Report on Emissions Scenarios (SRES) (http://ipcc-ddc.cru.uea.ac.uk/asres/sres_home_climate.html). Our predicted estimates of sea level rise, 30 cm to 100 cm by the year 2100, are based on SRES A1 which assumes rapid economic growth, low population growth and rapid introduction of new and more efficient technology.

SLAMM was developed with EPA funding in the mid 1980s (Park et al. 1986), and SLAMM2 was used to simulate 20% of the coast of the contiguous United States for the EPA

Report to Congress on the potential effects of global climate change (Park et al. 1989b, Park et al. 1989c) and subsequent summaries (Park 1991, Titus et al. 1991). Subsequently, more detailed studies were undertaken with SLAMM3, including simulations of St. Mary's Estuary, FL-GA (Lee et al. 1991, Lee et al. 1992, Park et al. 1991), Puget Sound (Park et al. 1993), and South Florida (Park and Lee 1993). More recently SLAMM4 was applied to all of San Francisco Bay, Humboldt Bay, and large areas of Delaware and Galveston bays (Galbraith et al. 2002, Galbraith et al. 2003).

SLAMM simulates the dominant processes involved in wetland conversions and shoreline modifications during long-term sea level rise. A complex decision tree incorporating geometric and qualitative relationships is used to represent transfers among coastal classes. Each site is divided into cells of equal area, and each class within a cell is simulated separately. Earlier versions of SLAMM used cells that were usually 500 by 500 m or 250 by 250 m. Version 4 uses cells that are 30 m by 30 m, based on NOAA tidal data, USFWS National Wetland Inventory data, and the USGS National Elevation Dataset that are readily available for downloading from the Web. Map distributions of wetlands are predicted under conditions of accelerated sea level rise, and results are summarized in tabular and graphical form.

Relative sea level change is computed for each site for each time step; it is the sum of the historic eustatic trend, the site-specific rate of change of elevation due to subsidence and isostatic adjustment, and the accelerated rise depending on the scenario chosen (Titus et al. 1991). Sea level rise is offset by sedimentation and accretion using average or site-specific values. For each time step the fractional conversion from one class to another is computed on the basis of the relative change in elevation divided by the elevational range of the class in that cell. For that reason, marshes that extend across wide tidal ranges are only slowly converted to unvegetated tidal flats. If a cell is protected by a dike or levee it is not permitted to change. The existence of these dikes can severely affect the ability of wetlands to migrate onto adjacent shorelines. Diked wetlands are assumed to be subject to inundation when relative sea-level change is greater than 2 m, although that assumption can be changed. In one study, alternate management scenarios involving maintenance of dikes were simulated (Park et al. 1993).

In addition to the effects of inundation represented by the simple geometric model described above, second-order effects occur due to changes in the spatial relationships among the coastal elements. In particular, the model computes exposure to wave action; if the fetch (the distance across which wind-driven waves can be formed) is greater than 9 km, the model assumes moderate erosion. If a cell is exposed to open ocean, severe erosion of wetlands is assumed. Beach erosion is modeled using a relationship reported by Bruun (1962, 1986) whereby recession is 100 times the change in sea level; that assumption can be changed if site-specific data are available. Wetlands on the lee side of coastal barriers are subject to conversion due to overwash as erosion of backshore and dune areas occurs and as other lowlands are drowned. Erosion of dry lands is ignored; in the absence of site-specific information, this could underestimate the availability of sediment to replenish wetlands where accelerated bluff erosion could be expected to occur. Coastal swamps and fresh marshes migrate onto adjacent uplands as a response of the water table to rising sea level close to the coast; this could be modified to take advantage of more site-specific predictions of water table elevations.

Predicted changes in marsh area and habitat type will be combined with habitat specific measurements of ecosystem services to quantify changes in delivery of regulation, habitat and production functions of tidal marshes under different scenarios of sea level rise. SLAMM will be employed to model incremental (10 cm) changes in sea level rise up to 1 m as well as extreme increases in sea level (1.5 m, 2 m) to identify thresholds of abrupt increases in submergence, habitat conversion and changes in ecosystem services.

Modeling results also will be used to assess changes in functions associated with disturbance regulation (i.e. shoreline protection) and food functions (commercial shrimp landings). The effect of rising sea level on disturbance regulation will be quantified by calculating an index of shoreline protection using marsh width and macrophyte stem density and diameter. Knutson (1988) developed a model that describes the reduction in wave energy as a function of tidal marsh width and stem density of *S. alterniflora*. Basically the model predicts the reduction in wave energy as a wave of height *a* passes through marsh width *w* (Knutson et al. 1982). Dissipation of wave energy depends on the density of stems (assumed to be cylindrical) and the distance of vegetated habitat that the wave translates through. Developed for Chesapeake Bay marshes, the model predicts that a marsh 30 m wide dissipates essentially of 100% of the energy of a 0.2 m wave. We will use our field-based measurements of stem density along with GIS measurements of changes in marsh width (caused by rising sea level) to predict how wave dissipation is altered under different scenarios of sea level rise.

The effects of sea level rise on commercial shrimp landing along the Georgia Coast also will be modeled using SLAMM4. Following the procedure of Park (1991), we will forecast the changes in brown and white shrimp catch statistics as a consequence of sea-level rise. First, a regression equation will be developed relating catch statistics for National Marine Fisheries Service statistical areas on the East Coast to the spatial complexity of adjacent salt marshes. The greater the complexity of the marsh-water interface, as represented by its fractal dimension, the greater the availability of habitat for shrimp nursery grounds. SLAMM computes the marsh-water fractal dimension, normalized to marsh area, and predicts its change as sea-level rise affects the extent of tidal creeks and marsh “ponds.” As Zimmerman et al. (1991) have observed and as SLAMM has predicted (Park 1991), sea-level rise can cause an increase in shrimp production during marsh breakup, followed by population crash as marsh habitat disappears (Figure 5).

SLAMM4 also will be used to predict the effects of dike maintenance and removal to protect marshes against rising sea level on tidal marsh area and delivery of ecosystem services. Ecosystem services of diked freshwater marshes will be combined with SLAMM-derived simulations of alternate management scenarios where (1) existing dikes are maintained and (2) existing dikes are breached to restore connectivity, tidal inundation and sediment deposition.

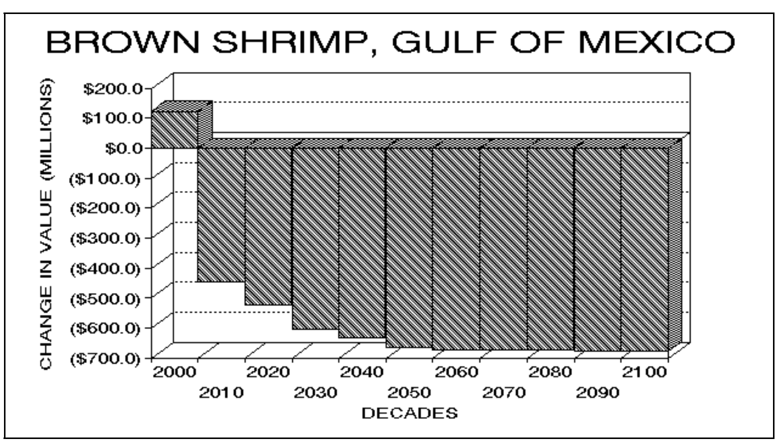


Figure 5. Predicted change in brown shrimp catch with a 1-mf sea-level rise by the year 2100 (Park 1991)

Results from the plot-level ecosystem services measurements will be scaled to the extent of the study region using the NWI-based SLAMM4 inputs, and the incremental SLAMM4 outputs, using a per unit area approach. The field and in situ study results will provide measurements of the ecosystem services that can be scaled to the level of plots, with known dimensions. The areas of each tidal wetland class are known for the SLAMM4 model (baseline) inputs and will be calculated for the SLAMM4 outputs associated with each incremental rise in sea level. This information enables the various ecosystem services to be directly scaled to the extent of the study region, changes in ecosystem service levels associated with different incremental rises in sea level to be calulated. Moreover, the incremental modeling approach will allow thresholds (i.e., inflection points) of change in ecosystem services associated with specific changes in sea-level to be identified, and applied to specific points in the future according to different anticipated rates of sea-level rise (i.e., scenarios).

Expected Results and Significance

Results from this study will be used to develop a model that describes the delivery of ecosystem services among different types of tidal marshes. To date, no systematic effort has been undertaken to understand how the kinds and magnitude of ecosystem services vary among salt, brackish and tidal freshwater marshes. Measurements to determine how ecosystem services respond to sea level rise at ecosystem- and landscape-scales will enable us to quantify (1) tidal wetland loss through submergence, (2) conversion of wetland habitat (i.e. replacement of tidal freshwater marsh by brackish and salt marshes) and (3) alteration in delivery of tidal marsh ecosystem services of the South Atlantic coast in response to different scenarios of climate change-induced sea level rise.

Permanent monitoring stations established at ten marshes of the GCE LTER study domain since 2000 enable us to evaluate the effects of inter-annual variability of temperature and precipitation on selected ecosystem services. The first two years of the LTER study (2000-2001) were characterized by drought conditions, with rainfall 70% of average followed by the return to more "normal" rainfall patterns in 2002 and 2003. Statistical and time series analysis of the permanent plot data in conjunction with of climate measurements (temperature, precipitation) and indices (salinity) will allow us to compare the delivery of selected ecosystem services between wet and dry years.

Deliverables from this study include (1) a conceptual model of ecosystem services provided by salt, brackish and tidal freshwater marshes, (2) changes in the area and delivery of ecosystem services, including commercial fisheries (shrimp) yields, of tidal marshes of the south Atlantic coast in response to rising sea level, (3) effects of maintaining existing dikes (to protect tidal freshwater marshes) versus removing dikes to restore connectivity on delivery of ecosystem services and (4) assessment of the effects of inter-annual climate variability (wet versus dry years) on tidal marsh ecosystem services.

In addition, the investigators are committed to providing for public *discovery* of the project and *accessibility* and *evaluation* of results. A project web-site, to be developed and hosted at Indiana University, will include a project description, workplan, results as they become available, links to the researchers, and acknowledgement of the EPA and STAR program including display of appropriate logo(s). The web site will feature a simple interactive web-mapping application through the geospatial project results (i.e., SLAMM output and scaled ecosystem service layers) are displayed with contextual information (e.g., administrative boundaries, roads, and geographic names). FGDC-compliant metadata will be created for all geospatial products, and the project and site will be registered via DIF-metadata in the Global Change Master Directory (GCMD). A project archive will be made available upon its conclusion.

General Project Information

Project Organization

Chris Craft (Indiana Univ.) will serve as the PI on the project. He will oversee coordination of field sampling, GIS & modeling. He also will be in charge of the soils & marsh nekton work. The Craft Lab is equipped with gamma spectrometer for ^{137}Cs and ^{210}Pb analysis, Perkin-Elmer CHN analyzer, UV/visible & atomic absorption spectrophotometers and wet lab space for soil digestions. The Craft Lab also will work to identify and statistically analyze marsh epifauna data from permanent plots in the ten GCE LTER marshes for years 2000-present.

Samantha Joye (Co-PI, Univ. of Georgia) will supervise the CO_2, CH_4 and denitrification analyses. Her group will be responsible for quantifying sediment fluxes of CO_2 and CH_4, for determining denitrification rates and for conducting laboratory experiments to determine the environmental and physiological factors controlling microbial activity in sediments.

Steve Pennings (Co-PI, Univ. of Houston) will supervise measurements of macrophyte production & diversity. He also will oversee statistical and time series analysis of climate and vegetation data from the GCE LTER permanent plots. Pennings is the Co-PI of the GCE LTER and will coordinate efforts between the LTER and this project.

Dick Park (Eco Modeling), who developed the SLAMM model, will refine and run the model for the three study watersheds and the Georgia-South Carolina coast. Associate Investigator Jeff Ehman (Pangaea Information Technologies Ltd.) will develop the GIS model that that is used to predict changes in marsh area, type and ecosystem services in response to vary scenarios of sea level rise and will provide geospatial data for the modeling effort.

All project PI's will communicate regularly with Craft, who will be responsible for overall project direction and coordination. In addition, all PI's will attend a one-day annual meeting, either in conjunction with field efforts in Georgia to discuss progress, future work and manuscripts.

Relationship to GCE LTER Project

This proposal will benefit from close collaboration with the GCE-LTER. GCE-LTER data sets have sparked many of our initial ideas, and, as described above, ongoing GCE-LTER monitoring will continue to provide data that will be critical to the success of this project. LTER sites are required to share data openly, and several of the PI's on this proposal are heavily involved with the GCE-LTER, ensuring easy access to the GCE-LTER data. The GCE-LTER will also, whenever possible, provide expertise, field and logistical support, and make equipment available to this project. At the same time, this project represents work that the GCE-LTER alone cannot do. The sampling scheme of the GCE-LTER (a grid of 10 marsh sites, most of which are fully saline) is inadequate to test the hypotheses raised in this proposal, because it covers only one major river (Altamaha River) and has no replication of fresh and brackish sites along that one river. In addition, GCE-LTER financial resources are committed to other projects, and there simply is not enough funding to support the work proposed here. Although this project will benefit substantially from close ties to the GCE-LTER, and could not be done without such an association (unless the budget were doubled), the work proposed here is outside the immediate focus of GCE-LTER research and will not be conducted without external funding.

References

Bartlett, KB, DS Bartlett, RC Harriss and DI Sebacher. 1988. Methane emissions along a salt marsh salinity gradient. *Biogeochemistry* 4:183-202.

Birch, JB and JL Cooley. 1982. Production and standing crop patterns of giant cutgrass (*Zizaniopsis milaceae*) in a freshwater tidal marsh. *Oecologia* 52:230-235.

Boumans, RMJ and JW Day, Jr. 1993. High precision measurements of sediment elevation in shallow coastal areas using a sedimentation-erosion table. *Estuaries* 16:375-380.

Bowden, WB. 1984. Nitrogen and phosphorus in the sediments of a tidal, freshwater marsh in Massachusetts. *Estuaries* 7:108-118.

Brinson, MM, RR Christian and LK Blum. 1995. Multiple states in the sea level induced transition from terrestrial forest to Estuary. *Estuaries* 18:648-659.

Brockmeyer, RE, JR Rey, RW Virnstein, RG Gilmore and L Earnest. 1997. Rehabilitation of impounded estuarine wetlands by hydrologic reconnection to the Indiana River Lagoon, Florida. *Wetlands Ecology and Management* 4:93-109.

Bruun, P. 1962. Sea level rise as a cause of shore erosion. *Journal of the Waterways and Harbors Division, Proceedings of the America Society of Civil Engineers* 88:117-130.

Bruun, P. 1986. Worldwide impacts of sea level rise on shorelines. Pp. 99-128. In, JG Titus (ed.) *Effects of Changes in Stratospheric Ozone and Climate Change, Vol. 4: Sea Level Rise*. U.S. Environmental Protection Agency. Washington, DC.

Cahoon, DR. 1994. Recent accretion in two managed marsh impoundments in coastal Louisiana. *Ecological Applications* 4:166-176.

Capone, DG and RP Kiene. 1988. Comparative microbial dynamics in marine and freshwater sediments: Contrasts in anaerobic carbon catabolism. *Limnology and Oceanography* 33:725-749.

Carlson, PR, LA Yarbro, FX Courtney, T Leary, H Arnold, D Leslie, J Hughes and N Craft. 2001. Panhandle salt marsh mortality: A prelude to Louisiana brown marsh? Abstract from *Coastal Marsh Dieback in the Northern Gulf of Mexico: Extent, Causes, Consequences and Remedies*. U.S. Geological Survey, Biological Resource Division, Information and Technology Report. USGS/BRD/ITR--2001-0003. Reston VA. 31 pp.

Chanton, JP and GJ Whiting. 1995. Chapter: "Trace gas exchange in freshwater and coastal marine systems" in Methods in Ecology: Trace Gases. Eds. P. Matson and R. Harriss. *Blackwell Scientific Publ.* pp. 98-125.

Childers, DL. 1994. Fifteen years of marsh flumes: a review of marsh-water column interactions in southeastern USA estuaries. Pp. 277-293. In W.J. Mitsch (ed.), *Global Wetlands: Old World and New*. Elsevier, Amsterdam.

Conner, RF, GL Chmura and CB Beecher. 2001. Carbon accumulation in Bay of Fundy salt marshes: Implications for restoration of reclaimed salt marshes. *Global Biogeochemical Cycles* 15:943-954.

Costanza, R, R d'Arge, R de Groot, S Farber, M Grasso, B. Hannon, K Limburg, S Naeem, RV O'Neill, J Paruelo, RG Raskin, P Sutton and M van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387:253-260.

Craft, CB, SW Broome and ED Seneca. 1988. Nitrogen, phosphorus and organic carbon pools in natural and transplanted marsh soils. *Estuaries* 11:272-280.

Craft, CB, ED Seneca and SW Broome. 1993. Vertical accretion in regularly and irregularly flooded microtidal estuarine marshes. *Estuarine Coastal and Shelf Science* 37:371-386.

Craft, CB, JM Reader, JN Sacco and SW Broome. 1999. Twenty five years of ecosystem development of constructed *Spartina alterniflora* (Loisel) marshes. *Ecological Applications* 9:1405-1419.

Craft, CB, JP Megonigal, S Broome, J.Stevenson, R Freese, J Cornell L Zheng and J Sacco. 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. *Ecological Applications* 13:1417-1432.

Dahl, TE and CE Johnson. 1991. *Status and Trends of Wetlands in the Conterminus United States, Mid-1970's to Mid-1980's*. U.S. Department of the Interior, Fish and Wildlife Service, Washington DC, 28 pp.

Daiber, FC. 1982. *Animals of the Tidal Marsh*. Van Nostrand Reinhold, New York.

Daily, GC, S Alexander, PR Ehrlich, L. Goulder, J Lubchenco, PA Matson, HA Mooney, S. Postel, SH Schneider, D Tilman and GM Woodwell. 1997. Ecosystem services: Benefits supplied to human societies by natural ecosystems. *Issues in Ecology*. No. 2, 1-16.

de Groot, RS, MA Wilson and RMJ Boumans. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* 41:393-408.

DeLaune, RD, CN Reddy and WH Patrick Jr. 1981. Accumulation of plant nutrients and heavy metals through sedimentation processes and accretion in a Louisiana marsh. *Estuaries* 4:328-334.

DeLaune, RD, A. Jugsujindi, GW Peterson and WH Patrick, Jr. 2003. Impact of Mississippi River freshwater reintroduction on enhancing marsh accretionary processes in a Louisiana Estuary. *Estuarine Coastal and Shelf Science* 58:653-662.

Doumlele, DG. 1981. Primary production and seasonal aspects of emergent plants in a tidal freshwater marsh. *Estuaries* 4:139-142.

Eddy, S. 1969. *The Freshwater Fishes*. Wm. C. Brown Company Publishers, Dubuque, Iowa.

Edwards, NT and JS Riggs. 2003. Automated Monitoring of Soil Respiration: A Moving Chamber Design. Soil Sci. Soc. Am. J. 67: 1266-1271.

Flory, J and M Alber. 2002. Dead marsh information. Georgia Coastal Research Council. November 2002. http://www.marsci.uga.edu/coastalcouncil/PDFs/marshsummarygcrc.pdf

GCRC (Georgia Coastal Research Council). 2004. *Proceedings of the Marsh Dieback Workshop*. February 3-4, 2004. Savannah Georgia.

Groffman, PM. 1994. Denitrification in freshwater wetlands. *Current Topics in Wetland Biogeochemistry* 1:15-35.

Hatton, RS, WH Patrick Jr and RD DeLaune. 1981. Sedimentation, nutrient accumulation and early diagenesis in Louisiana Barataria Basin coastal marshes. Pp. 255-267. In, VS Kennedy (ed.) *Estuarine Comparisons*. Academic Press, New York.

Hefner, JM, BO Wilen, TE Dahl and WE Frayer. 1994. *Southeast Wetlands, Status and Trends, Mid-1970's to Mid-1980's*. U.S. Department of the Interior, Fish and Wildlife Service, Atlanta GA, 32 pp.

Hester, MW, JM Willis, IA Mendelssohn, R Schneider and K McKee. 2004. Integrative approach to understanding the causes of salt marsh dieback: Determination of salt marsh species tolerance limits to potential environmental stressors and simulation of salt marsh dieback. In, *Proceedings of the Marsh Dieback Workshop*. Georgia Coastal Research Council. 2004. February 3-4, 2004. Savannah Georgia.

Hopkinson, CS, Jr. 1992. A comparison of ecosystem dynamics in freshwater wetlands. *Estuaries* 15:549-562

Hsieh, YP. 1996. Assessing aboveground net primary production of vascular plants in marshes. *Estuaries* 19:82-85.

Jenkins, WC and WM Kemp. 1984. The coupling of nitrification and denitrification in two estuarine sediments. *Limnology and Oceanography* 29:609-619.

Joye, S. B., 2002. Denitrification in the Marine Environment. In: Encyclopedia of Environmental Microbiology, G. Collins (Ed.), John Wiley & Sons, Inc., New York, pp. 1010-1019.

Joye, SB and JT Hollibaugh. 1995. Influence of sulfide inhibition of nitrification on nitrogen regeneration in sediments. *Science* 270:623-625.

Joye, S. B. and H. W. Paerl. 1994. Nitrogen cycling in microbial mats: rates and patterns of denitrification and nitrogen fixation. Mar. Biol. 199: 285-195.

Joye, SB, ML Mazzotta and JT Hollibaugh. 1996. Community metabolism in microbial mats: The occurrence of biologically-mediated iron and manganese reduction. Estuar. Coast. Shelf Sci. 43: 747-766.

Kana, TM, MB Sullivan, JC Cornwell and K Groszkowski. 1998. Denitrification in estuarine sediments determined by membrane inlet mass spectrometry. Limnol Oceanogr. 43: 334-339.

Karl, TR, RW Knight and N Plummer. 1995. Trends in high-frequency climate variability in the 20^{th} century. *Nature* 377:217-220.

Khan, H and GS Brush. 1994. Nutrient and metal accumulation in a tidal freshwater marsh. *Estuaries* 17:345-360.

Kneib, 1997. The role of tidal animals in the ecology of estuarine nekton. *Oceanography and Marine Biology: an Annual Review* 35:163-220.

Knutson, PA. 1988. Role of coastal marshes in energy dissipation and shore protection. Pp. 161-174. In, DD Hook and others (ed.), *The Ecology and Management of Wetlands, Vol. 1: Ecology of Wetlands*. Timber Press. Portland, Oregon.

Knutson, PA, RA Brochu, WM Seeling and M Inskeep. 1982. Wave dampening in *Spartina alterniflora* marshes. *Wetlands* 2:87-104.

Lee, DS, CR Gilbert, CH Hocutt, RE Jenkins, DE McAllister and JR Stauffer, Jr. 1980. *Atlas of North American Freshwater Fishes*. NC State Museum Natural History, Raleigh, NC.

Lee, J. K., R. A. Park, and P. W. Mausel. 1992. Application of Geoprocessing and Simulation Modeling to Estimate Impacts of Sea Level Rise on the Northeast Coast of Florida. *Photogrammetric Engineering and Remote Sensing* 58: 1579-1586.

Lee, J. K., R. A. Park, P. W. Mausel, and R. C. Howe. 1991. GIS-related Modeling of Impacts of Sea Level Rise on Coastal Areas. Pp. 356-367. *GIS/LIS '91 Conference,* Atlanta, Georgia.
Park, R. A. 1991. Global Climate Change and Greenhouse Emissions. Pages 171-182. *Subcommittee on Health and Environment, U.S. House of Representatives,* Washington DC.

Linthurst, RA. 1980. An evaluation of aeration, nitrogen pH and salinity as factors affecting *Spartina alterniflora* growth: A summary. Pp. 235-247. In, VS Kennedy (ed.), *Estuarine Perspectives*. Academic Press, New York.

Linthurst, RA and ED Seneca. 1980. Dieback of salt-water cordgrass (*Spartina alterniflora* Loisel) in the lower Cape fear River estuary of North Carolina: An experimental approach to reestablishment. *Environmental Conservation* 7:59-66.

Linthurst, RA and ED Seneca. 1981. Aeration, nitrogen and salinity as determinants of *Spartina alterniflora* Loisel growth response. *Estuaries* 4:53-63.

Mahlman, JD. 1997. Uncertainties in projections of human-caused climate warming. *Science* 278:1416-1417.

McKee, KL, IA Mendelssohn and MD. Materne. 2004. Acute salt marsh dieback in the Mississippi River deltaic plain: a drought-induced phenomenon? *Global Ecology and Biogeography* 13:65-73.

Merrill, JZ and JC Cornwell. 2000. The role of oligohaline marshes in estuarine nutrient cycling. Pp. 425-441. In, MP Weinstein and DA Kreeger (ed.) *Concepts and Controversies in Tidal Marsh Ecology*. Kluwer Academic Publishers. Dordrecht, Netherlands.

Michot, TC and C Wells. 2001. Statewide distribution of brown marsh as determined from aerial surveys. In, Stewart, RE, CE Proffitt and TM Charron. 2001. Abstract from *Coastal Marsh Dieback in the Northern Gulf of Mexico: Extent, Causes, Consequences and Remedies*. U.S. Geological Survey, Biological Resource Division, Information and Technology Report. USGS/BRD/ITR--2001-0003. Reston VA. 31 pp.

Moorhead, KK and MM Brinson. 1995. Response of wetlands to rising sea level in the lower coastal plain of North Carolina. *Ecological Applications* 5:261-271.

Morris, JT. 2001. The physical determinants of sediment salinity and their relationship to the brown marsh phenomenon in Louisiana. In, Stewart, RE, CE Priffitt and TM Charron. 2001. Abstract from *Coastal Marsh Dieback in the Northern Gulf of Mexico: Extent, Causes, Consequences and Remedies*. U.S. Geological Survey, Biological Resource Division, Information and Technology Report. USGS/BRD/ITR--2001-0003. Reston VA. 31 pp.

Morris, JT and Whiting, GJ. 1986. Emission of gaseous carbon dioxide from salt marsh sediments and its relation to other carbon losses. *Estuaries* 9, 9-19.

Morris, JT, B Kjerfve and JM Dean. 1990. Dependence of estuarine productivity on anomalies in mean sea level. *Limnology and Oceanography* 35:926-930.

Nixon, SW 1980. Between coastal marshes and coastal waters: a review of twenty years of speculation and research on the role of salt marshes in estuarine productivity and water chemistry. Pp. 437-525. In, P. Hamilton and K.B. McDonald (ed.), *Estuarine and Wetland Processes*. Plenum Press, New York.

NOAA (National Oceanic and Atmospheric Administration). 2000. *The Potential Consequences of Climate Variability and Change on Coastal Areas and Marine Resources*. DF Boesch, JC Field and D Scavia (ed.). NOAA's Coastal Ocean Program. Decision Analysis Series No. 21. U.S. Department of Commerce. Washington DC. 163 pp.

Odum, WE. 1988. Comparative ecology of tidal freshwater and salt marshes. *Annual Review of Ecology and Systematics* 19:147-176.

Odum, W.E., T.J. Smith, III, J.K. Hoover and C.C. McIvor. 1984. *The Ecology of Tidal Freshwater Marshes of the United States East Coast: A Community Profile*. FWS/OBS-84/17. U.S. Fish and Wildlife Service, Washington, DC.

Oremland, RS. 1988. Biogeochemistry of methanogenic bacteria. Pp. 641-705. In, AJB Zehnder (ed.), *Biology of Anaerobic Microorganisms*. John Wiley and Sons, New York.

Paludan, C and JT Morris. 1999. Distribution and speciation of phosphorus along a salinity gradient in intertidal marsh sediments. *Biogeochemistry* 45:197-221.

Park, RA. 1991. Global Climate Change and Greenhouse Emissions. Pages 171-182. *Subcommittee on Health and Environment, U.S. House of Representatives,* Washington DC.

Park, RA, TV Armentano, and CL Cloonan. 1986. Predicting the Effects of Sea Level Rise on Coastal Wetlands. Pp. 129-152. In, J. G. Titus (ed.) *Effects of Changes in Stratospheric Ozone and Global Climate, Vol. 4: Sea Level Rise*. U.S. Environmental Protection Agency, Washington, D.C.

Park, RA, JK Lee, and D Canning. 1993. Potential Effects of Sea Level Rise on Puget Sound Wetlands. *Geocarto International* 8: 99-110.

Park, RA, JK Lee, PW Mausel and RC Howe. 1991. Using remote sensing for modeling the impacts of sea level rise. *World Resources Review* 3:184-220.

Park, RA, MS Trehan, PW Mausel and RC Howe. 1989a. Coastal wetlands in the twenty first century: Profound alterations due to rising sea level. Pp. 71-80. In, *Wetlands: Concerns and Successes*. Proceedings of a Symposium sponsored by American Water Resources Association, Bethesda, MD.

Park, RA, MS Trehan, PW Mausel, and RC Howe. 1989b. The Effects of Sea Level Rise on U.S. Coastal Wetlands. Pp. 1-1 to 1-55. In, JB Smith and DA Tirpak (Ed.) *The Potential Effects of Global Climate Change on the United States, Appendix B - Sea Level Rise*. U.S. Environmental Protection Agency, Washington, D.C.

Park, R A, MS Trehan, PW Mausel, and RC Howe. 1989c. *The Effects of Sea Level Rise on U.S. Coastal Wetlands and Lowlands*. Pages 48 pp. + 789 pp. in appendices. Holcomb Research Institute, Butler University, Indianapolis, Indiana.

Perry, JE and RB Atkinson. 1997. Plant diversity along a salinity gradient of four marshes on the York and Pamunkey Rivers in Virginia. *Castanea* 62:112-118.

Peterson, BJ, RW Howarth and RH Garritt. 1985. Multiple stable istopes used to trace the flow of organic matter in estuarine food webs. *Science* 227:1361-1363.

Phleger, CF. 1971. Effect of salinity on growth of a salt marsh grass. *Ecology* 52:908-911.

Portnoy, J. W. (1999) “Salt Marsh Diking and Restoration: Biogeochemical Implications of Altered Wetland Hydrology” Environmental Management 24:111 – 120.

Rey, JR, RA Crossman, T Kain and J Shaffer. 1991. Surface water chemistry of wetlands and the Indian River Lagoon, Florida. *Journal of the Florida Mosquito Control Association* 62:25-36.

Rey, JR, J Shaffer, D Tremain, RA Crossman and T Kain. 1990. Effects of re-establishing tidal connections in two impounded tropical marshes on fishes and physical conditions. *Wetlands* 10:27-47.

Richardson, CJ. 1994. Ecological functions and human values in wetlands: A framework for assessing forestry impacts. *Wetlands* 14:1-9.

Sacco, JN, ED Seneca and T Wentworth. 1994. Infaunal community development of artificially established salt marshes in North Carolina. *Estuaries* 17:489-500.

SAS (Statistical Analysis Systems). 1996. *SAS/Stat User’s Guide*. SAS Institute Inc., Cary, NC.

Schlesinger, WH. 1997. *Biogeochemistry: An Analysis of Global Change*. Academic Press, New York.

Schubauer, JP and CS Hopkinson. 1984. Above- and belowground emergent macrophyte production and turnover in a coastal marsh ecosystem. *Limnology and Oceanography* 29:1052-1065.

Seitzinger, SP. 1988. Denitrification in freshwater and coastal marine ecosystems: Ecological and Geochemical significance. *Limnology and Oceanography* 33:702-724.

Simpson, RL, RE Good, MA Leck and DF Whigham. 1983. The ecology of freshwater tidal wetlands. *BioScience* 33:255-259.

Sinicrope, TL, PG Hine, RS Warren and WA Niering. 1990. Restoration of an impounded salt marsh in New England. *Estuaries* 13:25-30.

Smith, WG. 1970. Spartina ‘dieback’ in Lousiana marshes. Coastal Studies Bulletin No. 5. Louisiana State University. Baton Rouge, Louisiana. Pp. 89-95.

Sokal, R.R. and F.J. Rohlf. 1995. *Biometry*. W.H. Freeman and Co., San Francisco.

Stewart, RE, CE Priffitt and TM Charron. 2001. Abstract from *Coastal Marsh Dieback in the Northern Gulf of Mexico: Extent, Causes, Consequences and Remedies*. U.S. Geological Survey, Biological Resource Division, Information and Technology Report. USGS/BRD/ITR--2001-0003. Reston VA. 31 pp.

Stocks, KI and JF Grassle. 2003. Benthic macrofaunal communities in partially impounded salt marshes in Delaware: Comparisons with natural marshes and responses to sediment exposure. *Estuaries* 26:777-789.

Sturdevant, A., C.B. Craft and J.N. Sacco. 2002. Effects of impoundment on ecological functions of estuarine marshes along Woodbridge River, NY/NJ harbor. *Urban Ecosystems* 6:163-181.

Titus, J. G., R. A. Park, S. P. Leatherman, J. R. Weggel, M. S. Greene, P. W. Mausel, M. S. Trehan, S. Brown, C. Grant, and G. W. Yohe. 1991. Greenhouse Effect and Sea Level Rise: Loss of Land and the Cost of Holding Back the Sea. *Coastal Management* 19: 171-204.

Whigham, DF, J McCormick, RE Good and RL Simpson. 1978. Biomass and primary production in tidal freshwater wetlands of the middle atlantic coast. Pp. 3-20. In, RE Good, DF Whigham and RL Simpson (ed.) *Freshwater Wetlands: Ecological Processes and Management Potential*. Academic Press, New York.

Wiegert, RG and BR Freeman. 1990. *Tidal salt marshes of the southeastern Atlantic coast: a community profile*. Biological report 85(7.29). U.S. Dept. of Interior, Fish and Wildlife Service. Washington D.C.

Williams, AB. 1984. Shrimp, lobsters and crabs of the Atlantic coast of the eastern United States, Maine to Florida. Smithson. Inst. Press, Washington, DC.

DOE National Institute for Climatic Change Research (NICCR)

Notice RFP-03

Summary

The U.S. Department of Energy (DOE) National Institute for Climatic Change Research (NICCR) hereby announces its interest in receiving research proposals. Proposed research is requested that would improve understanding of potential effects of contemporary climatic change on the structure and functioning of important terrestrial ecosystems within the United States, as well as possible feedbacks from terrestrial ecosystems to climate and atmospheric composition. NICCR divides terrestrial ecosystems into two groups: inland (not adjacent to an ocean) and coastal (adjacent to an ocean, including barrier islands).

For inland terrestrial ecosystems, the main climatic changes of interest are changes in temperature and precipitation. Research should: (1) reduce scientific uncertainty about potential effects of climatic change on the structure and functioning of terrestrial ecosystems; (2) evaluate or improve the understanding and prediction of potential effects of climatic change on the future geographic distribution of terrestrial ecosystems at the regional scale; (3) use measurements of contemporary exchanges of mass and energy between the atmosphere and terrestrial ecosystems to reduce scientific uncertainty about possible effects of an altered terrestrial carbon cycle and/or surface energy exchange on global and/or regional climate; or (4) use synthesis of existing experimental or observational data, or modeling, to better understand or forecast potential effects of climatic change on ecological systems and/or feedbacks from terrestrial ecosystems to climate at the regional scale.

For coastal ecosystems, the climatic changes of interest are sea-level rise and the possibility of increased frequency and/or intensity of storms (including hurricanes) directly affecting coastal ecosystems. Ecosystems to be studied will be the terrestrial ecosystems (including wetland and freshwater ecosystems, but not marine or estuarine ecosystems) that could be directly and significantly altered by sea-level rise or increased frequency or intensity of coastal storms. The ecological endpoints of interest are ecosystem or species migrations, changes in biodiversity, changes in primary production, or alterations in goods and services uniquely supplied by coastal terrestrial ecosystems.

Eligibility

United States colleges, universities, and not-for-profit, non-governmental research institutions are eligible for support through NICCR. The Principal Investigator must be principally employed by an eligible institution. Contractors to and employees of federal facilities, including federal agency laboratories and Federally Funded Research and Development Centers (FFRDCs), are ineligible for support. Subcontracts to ineligible institutions (e.g., government agencies, laboratories, or facilities and FFRDCs) will not be allowed. Questions about eligibility should be directed to Dr. Jeff Amthor (see Contact Persons below).

Projects funded through NICCR will have one Principal Investigator, except in the case of Collaborative Projects, as defined below. Principal Investigators may submit only one preproposal and a scientist cannot concurrently be Principal Investigator on more than one NICCR project or award.

Dates

Preproposals are REQUIRED. Full proposals will only be accepted from applicants who: (1) submit a compliant preproposal on time and (2) are informed by NICCR that their preproposal was selected to be developed into a full proposal.

Preproposals are due 5:00 PM Pacific Time, May 15, 2007.
Proposals are due 5:00 PM Pacific Time, August 21, 2007.

Research project start dates of about April 1, 2008, are expected.

Application Materials

Preproposals and proposals must be submitted electronically (uploaded) to the NICCR web site (http://niccr.nau.edu). Proposals should be contained within a single pdf file. The pdf file should be no larger than 3 MB. The required preproposal and proposal formats are described below in the sections "Preproposal Submission and Format" and "Proposal Format". Required templates for the Cover Page and Budget Page(s) are available for download at the NICCR web site.

Organization of NICCR

The NICCR is composed of four Regional Centers (encompassing all 50 states and the District of Columbia) and one Coastal Center. Research on inland terrestrial ecosystems is managed by the four Regional Centers while research on coastal terrestrial ecosystems is managed by the Coastal Center.

For inland research projects, the states are distributed among the four NICCR regions as follows:

Northeastern Region -- Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, Connecticut, New York, New Jersey, Pennsylvania, Delaware, Maryland, West Virginia, the District of Columbia, and Virginia.

Southeastern Region -- Texas, Louisiana, Arkansas, Mississippi, Alabama, Florida, Georgia, South Carolina, North Carolina, Tennessee, and Kentucky.

Midwestern Region -- North Dakota, South Dakota, Nebraska, Kansas, Oklahoma, Missouri, Iowa, Minnesota, Wisconsin, Illinois, Indiana, Ohio, and Michigan.

Western Region -- Alaska, Hawaii, Washington, Oregon, California, Idaho, Nevada, Arizona, Utah, Montana, Wyoming, Colorado, and New Mexico.

All states with a seashore are within the scope of Coastal Center research interests.

Contact Persons

Northeastern Region: Dr. Ken Davis, (814) 863-8601, davis@met.psu.edu

Southeastern Region: Dr. Rob Jackson, (919) 660-7408, jackson@duke.edu

Midwestern Region: Dr. Kurt Pregitzer, (906) 487-2396, kspregit@mtu.edu
(after June 15, 2007: Dr. Andrew Burton, (906) 487-2566, ajburton@mtu.edu)

Western Region: Dr. Bruce Hungate, (928) 523-0925, bruce.hungate@nau.edu

Coastal Center: Dr. Torbjörn Törnqvist, (504) 314-2221, tor@tulane.edu

Eligibility: Dr. Jeff Amthor, (301) 903-2507, jeff.amthor@science.doe.gov

Background

The purpose of NICCR is to mobilize university scientists, from all regions of the country, in support of the research goals of DOE's Climate Change Research Division (in the DOE Office of Science's Office of Biological and Environmental Research). Information about the Division's research goals is at http://www.sc.doe.gov/ober/CCRD_top.html).

The NICCR national web site is at http://niccr.nau.edu. Web sites maintained by the NICCR Centers are linked to the national web site.

Request for Proposals

This notice solicits proposals to conduct research related to effects of climatic changes on terrestrial ecosystems and potential feedbacks from terrestrial ecosystems to climate and atmospheric composition. Proposals should state clearly how the proposed research will fill important knowledge gaps that hinder regional-scale or national-scale forecasts of effects of climatic change on important ecosystems or feedbacks from ecosystems to climate. Proposals should state why the ecosystems to be studied are important. Criteria that could be considered are areal extent, primary production relative to total production in the region, habitat for threatened and endangered species, or ecosystem characteristics that would allow extrapolation of results to large areas or many ecosystem types.

It is expected that most projects supported by NICCR will be for individual investigators or small research teams (single institutions), but coordinated, multi-institutional (collaborative) projects will be considered for funding (see Collaborative Projects section). The purpose of considering collaborative projects is to allow science questions to be addressed that cannot be readily addressed with traditional single-investigator research projects, and to encourage synthesis activities that are integrated into observational or experimental studies at the regional scale; collaborative proposals should clearly state how their collaborative nature will satisfy these objectives. Proposals for Collaborative Projects must present either evidence of past collaborative success, or a clear plan that will facilitate successful project integration.

Only proposals addressing one of the following five focus areas will be considered for support in response to this Notice. The first four foci are associated with inland terrestrial ecosystems and will be administered by the four Regional Centers. The fifth focus is associated with coastal ecosystems and will be administered by the Coastal Center. Research addressing all foci should be directed at climatic changes possible during the next 50-100 years in the United States, and all research will be conducted within the United States.

<u>Focus 1</u>: Focus 1 projects will address potential effects of climatic change on terrestrial ecosystems. Projects should determine the theoretical and/or empirical basis of whether, and how, changes in temperature (annual mean temperature, seasonal and/or diel temperature cycles) and/or changes in precipitation (annual amount, number of events, temporal distribution of events and amounts) might affect the structure and functioning of important U.S. terrestrial ecosystems. Such research could include consideration of threshold effects of extreme temperature and/or precipitation "events" or periods (e.g., heat waves, extended droughts, extended wet periods, or significantly altered snowpack) on terrestrial ecosystems. This objective should be met through manipulative experiments in the field or laboratory (field research will generally be given higher priority during project selection). Experiments could be (1) entirely new or (2) value-added additions to ongoing experiments. Experimental manipulations of temperature and/or precipitation could include other "climatic change" factors, such as elevated CO_2 concentration. To the extent that "model" or "constructed" ecosystems can be justified for the study of ecosystem structure and functioning, experiments using such systems will be considered for support. Experimental research based on underlying theory would be especially relevant. The magnitude of temperature and/or precipitation manipulations should be clearly justified.

Ecological endpoints of interest include changes to (a) net primary production, (b) ecosystem-scale species composition and diversity, (c) ecosystem-atmosphere energy exchange (including albedo), and (d) ecosystem susceptibility to pests and disturbances. Proposed research should be directed at measurable and specified endpoints attainable within the proposed project period. Proposals should state briefly and clearly how the research results could or will be used to improve models of terrestrial ecosystems relevant to climatic change issues.

Focus 2: Focus 2 projects will improve the scientific basis for detecting or projecting changes in the geographic boundaries of U.S. terrestrial ecosystems (or biomes), and the populations of their dominant plant or animal species, in response to potential climatic changes. This goal could be achieved through improved understanding of changes to abundance, growth, reproduction, dispersal, or competitive interactions of major plant or animal species within their present ranges that might eventually lead to changes in the species composition or geographic distribution of communities. Climatic changes of interest are the annual mean and seasonal and diel cycles of temperature and the annual amount, frequency, and temporal distribution of precipitation (and available soil moisture). Studies of the potential effects of these climatic changes on important ecosystem disturbances (e.g., changes in the frequency or areal extent of fires) might be appropriate. The magnitude of climatic changes to be studied, and any relationships to major ecosystem disturbances, should be clearly justified.

The intent of this research is to reduce scientific uncertainty about how the geographic distribution of U.S. terrestrial ecosystems, and their component organisms, might be altered by future climatic changes or how those changes in the distribution of ecosystems might in turn alter regional climate. Projects might also attempt to determine if climatic changes during the past 100 years resulted in population and/or ecosystem movements. A particular emphasis should be placed on the relationship between geographic distributions of terrestrial ecosystems and their dominant organisms as that relationship is affected by climate. An important objective is to advance the development and evaluation of models of regional (of the order of one million square kilometers), national, or global biogeography that are, or may be, used to project effects of climatic change on the geographic distributions of terrestrial organisms and ecosystems in the United States. Toward this goal, all proposals should explain clearly how the research results will help develop, improve, or evaluate relevant models.

Projects using existing models of biogeography to make predictions of effects of specified climatic change scenarios on the future distribution of terrestrial ecosystems without accompanying improvements to those models, or directed efforts at evaluating the usefulness or accuracy of those models, will not be considered for support.

Focus 3: Focus 3 projects will address the measurement and analysis of contemporary exchanges of mass and energy between the atmosphere and terrestrial ecosystems or landscapes, and the use of those measurements and analyses to evaluate mechanisms that are, or that might be, included in climate and carbon cycle models. The intent of this research is to reduce scientific uncertainty about the potential effects of climatic change on atmosphere-ecosystem exchanges of mass and energy with an emphasis on the carbon cycle. Projects should use appropriate methods (including, but not limited to, eddy covariance) to quantify and understand ecosystem-atmosphere exchanges relevant to the climate system, focusing on regionally important terrestrial ecosystems (e.g., those covering significant area of land). Proposals should state how the research will improve understanding of the role of terrestrial ecosystems in regional or global cycles of energy and mass, with a focus on the biological control of those cycles and the effects of climatic change on that control.

Examples of relevant studies could include but are not limited to: (a) evaluation of specific ecological processes that may be either poorly represented or altogether lacking in climate and carbon cycle models, (b) efforts to detect changes in ecosystem functioning caused by recent climatic changes, or (c) efforts to improve observational methods essential to improving our ability to predict the effects of climatic change on ecosystem-atmosphere carbon and energy exchanges. Studies can be new observational programs, continuations of existing measurements, or additions to ongoing observational programs. Though proposals will be evaluated primarily on the hypotheses to be addressed in response to this RFP, the value of observations to the broader scientific community (e.g., value of the data to complementary, ongoing research projects; relevance to a broader network of measurements) will be considered. Such broader relevance should still fit within the foci of this RFP.

Key endpoints of the proposed research should include (a) improved quantitative understanding of the importance of terrestrial ecosystems as sources and sinks of greenhouse gases, leading to an improved ability to predict how those sources and sinks might change in response to climatic change during the coming 50–100 years; (b) improved quantitative understanding of how the surface energy balance of terrestrial ecosystems might change in response to climatic change during the coming 50-100 years; and/or (c) reduced uncertainty in future climatic change as a result of improved quantification of these potential feedbacks.

Focus 4: Focus 4 projects will carry out synthesis activities (including, but not limited to, process modeling) related to effects of climatic variability and change on U.S. terrestrial ecosystems, or feedbacks from terrestrial ecosystems to climatic change, principally with a regional focus. Projects should synthesize and advance mechanistic understanding of how future climatic variability and change might influence terrestrial ecosystem structure and functioning. Projects that include an effort to identify and quantify important scientific knowledge gaps or integrate multiple sources of information (e.g., observations and experiments) are encouraged. Short-term (e.g., one year) synthesis activities that can address a clearly defined gap in scientific knowledge are eligible for funding, and are encouraged.

The spatial scale of the research should be encompassed within, or up to the size of, a NICCR region. Synthesis could involve: (1) development of new, or use of existing, databases; (2) development and/or evaluation of ecological models; (3) meta-analysis; and/or (4) other appropriate research activities focused on advancing fundamental understanding of how and why climatic variability and change might influence the structure, functioning, and geographic distribution of terrestrial ecosystems at the regional scale. Projects might concentrate on: (a) interactions among climatic variability/change and disturbances (natural or human-caused); (b) effects of multiple changes in climate and atmospheric composition (e.g., ecological effects of concomitant changes in temperature, precipitation, and [CO_2]); or (c) detection, prediction, and/or modeling of regional-scale feedbacks between climatic change and terrestrial ecosystem functioning, including energy and greenhouse gas exchanges (i.e., feedbacks to the climatic system).

Research directed at climatic change mitigation options, including carbon sequestration in terrestrial ecosystems, will not be considered for support.

Focus 5: Focus 5 projects will reduce scientific uncertainty about potential effects of climatic change on coastal ecosystems in the United States. The environmental changes of interest are sea-level rise (which in some cases will be affected by coastal subsidence) and the possibility of increased intensity and/or frequency of storms, including hurricanes. Ecosystems to be studied will be the terrestrial ecosystems located on or very near the coast (ranging from mean sea level to the landward limit of likely direct and significant effects of sea-level rise and/or coastal storms in the next century), including ecosystems on barrier islands. Open aquatic ecosystems are excluded. Research may involve manipulative experiments (in the field or the laboratory as appropriate), mechanistic modeling, paleoenvironmental analyses, or observational studies, including remote sensing studies. All studies must be relevant to improved understanding of potential ecological effects of sea-level rise and/or changes in the intensity or frequency of coastal storms during the coming 50-100 years. The magnitude of sea level rise and/or altered frequency or intensity of storms to be studied should be clearly justified.

Ecological endpoints of interest include changes in the geographic (spatial) boundaries of whole ecosystems or communities, ecosystem-scale species composition and biodiversity, net primary production, and the goods and services supplied by coastal ecosystems.

Specific questions that could be addressed include: (1) How might coastal terrestrial ecosystems and their services to society be affected by changes in relative sea level and frequency or intensity of storms, including hurricanes? (2) What are the ecological mechanisms of species or community responses to sea level rise and related coastal environmental changes? (3) Can past climatic and sea-level changes as recorded by coastal-wetland sediments be used to improve understanding of potential future changes in coastal wetlands caused by climatic change and sea-level rise?

Synthetic projects combining aspects of experimental manipulations, simulation modeling, and observational studies (including remote sensing) are encouraged.

All Research Foci (Foci 1–5): All proposals should briefly and clearly describe tangible outcomes (products or deliverables) of the proposed activities, including a timetable of those outcomes. It is expected that all data and analyses (including model codes) obtained and developed with NICCR support will be made available to the public in a timely manner. Each proposal must *briefly* state how and when the data and analyses (including model codes) will be made publicly available and must include sufficient resources to provide this access. For example, data collected, databases synthesized, and/or codes developed might be posted to a public web site within 12 months of collection (for data) or development (for syntheses conducted and codes developed). Public access will not circumvent the rights of investigators to publish their work and/or receive proper acknowledgement for use of their research products. Established data and metadata formats should be used when they exist.

Collaborative Projects

A Collaborative Project is defined as one that involves substantial NICCR support to more than one institution. Specifically, any project for which more than 25% of the total budget request is from each of two or more institutions will be considered a Collaborative Project. (Projects involving a modest subcontract to a second institution, i.e., less than 25% of the total project budget, are not classified by NICCR as a Collaborative Project.) The potential collaborators should choose a single Principal Investigator and institution for the purpose of submitting a preproposal.

For successful preproposals, a single collaborative proposal should be submitted by one of the Principal Investigators/institutions. That single proposal should include a cover page for each of the collaborative institutions. Each collaborating institution must identify a single Principal Investigator from that institution. Those Principal Investigators must be listed as Co-Investigators on the proposal Cover Pages from each of the other institutions.

For successful Collaborative Project proposals, separate awards will be made to each of the collaborating institutions.

Funding

Annual budgets for individual projects (both inland and coastal) are expected to not exceed $125,000, unless there is prior approval obtained at the preproposal stage for more costly manipulative studies or larger collaborative (multi-institutional) projects.

<u>Inland ecosystems</u>. It is expected that about $2 million will be available in 2008 for new research on inland terrestrial ecosystems (i.e., up to about 16 new projects), contingent on availability of appropriated funds. Those funds will be divided about equally among the four Regional Centers. Out-year support will be contingent on availability of funds, progress of research, and DOE programmatic needs.

<u>Coastal ecosystems</u>. It is expected that about $0.7 million in 2008 will be available for new research on coastal terrestrial ecosystems (i.e., up to about six new projects), contingent on availability of appropriated funds. Out-year support will be contingent on availability of funds, progress of research, and DOE programmatic needs.

Publication of research results must acknowledge NICCR support as follows. “This research was supported by the U.S. Department of Energy’s Office of Science (BER) through the [Coastal; Northeastern Regional; Midwestern Regional; Southeastern Regional; or Western Regional] Center of the National Institute for Climatic Change

Research at [Tulane; The Pennsylvania State; Michigan Technological; Duke; or Northern Arizona] University."

Merit Review

Each proposal will be reviewed for technical merit by at least three independent reviewers. Reviewers will evaluate the scientific merit of the proposed research, the appropriateness of the proposed methods, the competency of the research team, and the appropriateness of the proposed budget. Proposals will also be evaluated by NICCR, for their relevance to the terms of this Notice and with respect to the balance of research projects within NICCR, including the balance of research topics within the NICCR regions. The DOE will evaluate the proposals for their relevance to the goals of the DOE climatic change research program.

Preproposal Submission and Format

Preproposals are REQUIRED. Preproposals must be prepared using the preproposal template available at the NICCR web site (http://niccr.nau.edu). Background material (literature review) should not be included in the preproposal. Preproposals must indicate whether a project is to be a Collaborative (multi-institutional) Project (see the preproposal template).

For inland research, preproposals should be submitted to the Regional Center in which the majority of the research will be conducted, which could be a different region than that containing the PI's institution. For all coastal research, preproposals should be submitted to the Coastal Center.

All preproposals will be reviewed and evaluated for scientific merit and adherence to the terms of this Notice by an independent panel. Adherence to the terms of this RFP will constitute 50% of the evaluation. Recommendations of that panel will then be reviewed by NICCR, which will make final decisions about the preproposals. Those decisions will be communicated to the applicants via e-mail as soon after the submission deadline as possible. Written reviews of (or comments on) the preproposals will not be provided by NICCR.

Proposals will be accepted only from applicants who are notified by NICCR that their preproposal was selected to be developed into a full proposal.

Proposal Format

The proposal must be prepared with one-inch margins all around, and no more than 51 lines of text per page (11 point font, single spacing). (The margins of the cover page are set by the cover page template; they are less than one inch.) Text in tables, figures, and figure legends can be more compact, but must be legible. The entire proposal should be contained within a single pdf file (maximum size of 3 MB) uploaded at the NICCR web site (http://niccr.nau.edu).

The proposal should include (in this order):
Cover Page(s)
Budget(s) and Budget Explanation(s)
Abstract (on a page by itself)
Narrative (15 pages maximum)
Literature Cited
Biographical Sketch(es)
Other Support of Investigator(s) (including project abstracts for related projects, see below)
Annotated Bibliography of Prior NIGEC or NICCR Research (if applicable; see below)

Cover Page: The required template for the cover page is available at the NICCR web site (http://niccr.nau.edu; see Cover Page Format). It should be downloaded from the web site. All fields indicated on the cover page template must be completed, except the co-investigator block, which is to be used only for Collaborative Projects.

For Collaborative Projects, a separate Cover Page for each collaborating institution must be included. These should all be placed at the top of the proposal.

Budget and Budget Explanation: The required template and instructions for the budget are available at the NICCR web site (http://niccr.nau.edu; see Budget Page and Instructions). There should be one budget page for each year (12-month period) of the proposed project. The budget pages must be followed by a budget explanation (using a separate budget justification for each year, with the justification for each year beginning on a new page) including a brief description of each budget item and a justification (or explanation) for the requested amounts. This should include brief descriptions of tasks to be performed by each named person to be funded by NICCR and uses to be made of any purchased items. Any travel should be described. Any subcontracts should be sufficiently described in the budget justification. If any single subcontract exceeds $25,000 during any year, separate budget page(s) and separate budget explanation pages must be included for that subcontract.

For Collaborative Projects, separate budget pages must be included for each of the collaborating institutions.

The budget explanation must be followed by a **copy of a signed agreement that establishes the allowable rates of reimbursement entered into between the Principal Investigator's employing institution and the U.S. Government (Federally audited and approved)**. For Collaborative Projects, a copy of the signed agreement for each of the collaborating institutions must be included. This agreement should not, however, be included in proposals submitted from scientists at the NICCR host university.

Abstract (on a page by itself): At the top of the abstract give the project title. On separate lines list the name(s) and employing institution(s) of the Principal Investigator and any coinvestigator(s). The text of the Abstract should be 400 words or less. It should contain five paragraphs as follows: (1) a statement, in broad scientific terms, of the project objectives; (2) a list of the specific hypotheses to be tested or science questions to be answered; (3) a statement of the location(s) of the research activities; (4) a brief outline, in general terms, of the approach (methods) to be used; and (5) a statement of what the research is intended to accomplish, including expected deliverables.

<u>Narrative</u>: The narrative comprises the research plan for the project and is limited to **15 pages (maximum)**. It should contain enough background material in the Introduction, including review of the relevant literature, to demonstrate sufficient knowledge of the state of the science and to set the stage for the proposed research. The major part of the narrative should be devoted to a description and justification of the proposed project, including details of the methods to be used. It should also include a timeline for the major activities of the proposed project, and should indicate which project personnel will be responsible for which activities. If any portion of the project is to be done in collaboration with another institution (or institutions), the narrative must provide information on the institution(s) and what part of the project it will carry out. Further information on any such arrangements is to be given in the sections “Budget and Budget Explanation”.

If the Principal Investigator or co-investigator(s) are supported by another DOE program to perform similar research, the narrative must state (no more than one-half page) how this proposed project differs from the other DOE-funded research being conducted. If the Principal Investigator or co-investigator(s) are (or were) supported by NIGEC or NICCR to perform related research, a brief description of the results of that research, and the length of NIGEC support, must be included in the Narrative in a section titled “Results of Related NIGEC/NICCR Research” (two pages maximum). This section of the Narrative (if included) should reference an Annotated Bibliography to be placed at the end of the proposal (see below).

<u>Literature Cited</u>: Give full bibliographic citations for all literature cited in the Narrative.

<u>Biographical Sketch(es)</u>: Biographical sketches of all senior personnel must be included and are limited to two pages each. Each sketch should include, at the end, a list of all collaborators and co-authors during the past 48 months, to be used to determine potential conflicts of interest when selecting reviewers.

<u>Other Support of Investigator(s)</u>: For each of the senior personnel, a list of current and pending research support should be included. An abstract of any current research project that is related to the proposed NICCR research project (in scope or location) must be included (no more than one half page per project/abstract).

<u>Annotated Bibliography of Prior NIGEC or NICCR Research</u> (if required; see Narrative above): The annotated bibliography is a list of publications resulting from prior NIGEC research, with each citation followed by a brief--no more than 150 words--description of the publication and its scientific importance. The purpose of the annotation is to demonstrate the relevance of these publications of previous NIGEC-funded research to

the stated goals of the previous NIGEC or NICCR project as well as the broad significance of the results.

Hydrogeomorphic Control of Nutrient and Sediment Removal by Freshwater Wetlands.

Christopher Craft[1], Joe Schubauer-Berigan [2]

[1]School of Public and Environmental Affairs, Indiana University, Bloomington IN 47405
[2]National Risk Management Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati OH 45268

We surveyed the literature to evaluate the role of landscape position, hydrologic connectivity, loading rate and wetland age on nitrogen (N) and phosphorus (P) removal by freshwater wetlands. Nitrogen and P removal is three times greater in connected (floodplain, fringe) wetlands than in depressional wetlands. In floodplain wetlands, 8-15 MT N/km^2 and 1-3 MT P/km^2 are sequestered annually in soil as compared to 3 MT N/km^2/yr and 0.5 MT P/km^2/yr for depressional wetlands. Denitrification removes an additional 3 to 15 MT of N/km^2/yr under low nitrate loadings. Nitrogen removal is stimulated by increased nutrient loading, mostly through greater denitrification, and, in highly loaded wetlands, N removal may exceed 10-50 MT/km^2 wetland/yr. Increased nutrient loading also boosts P removal though P removal (1-5 MT/km^2/yr) is an order of magnitude less than N. And P removal declines with time as sedimentation reduces water storage capacity and sorption sites become saturated. Creation, restoration and enhancement of wetlands for nutrient and sediment removal must recognize that (1) not all wetlands are equal when it comes to nutrient removal, (2) N removal is greater than P removal and (3) effective N removal is sustainable over time but P removal declines as wetlands age.

Contact Information: Christopher Craft, School of Public and Environmental Affairs, Indiana University, Bloomington IN 47405 USA
Phone: 812 855-5971, Fax: 812 855-7802, Email: ccraft@indiana.edu

Limnol. Oceanogr., 52(3), 2007, 1220–1230

Freshwater input structures soil properties, vertical accretion, and nutrient accumulation of Georgia and U.S. tidal marshes

Christopher Craft[1]
School of Public and Environmental Affairs, Indiana University, Bloomington, Indiana 47405

Abstract

To identify relationships between freshwater input and marsh soil properties, measurements of bulk density, nutrients (carbon [C], nitrogen [N], phosphorus [P]), accretion, and accumulation were compared in tidal marshes of three estuaries of Georgia that varied in delivery of freshwater. Soil organic C and N (0–30 cm) were two times greater in marshes of the freshwater-dominated Altamaha River than in the salt marshes of Doboy Sound and Sapelo River. ^{137}Cs accretion and accumulation of organic C and N were three to five times greater in freshwater-dominated marshes than in salt marshes. The patterns observed in Georgia marshes were geographically general; data for tidal freshwater and brackish marsh soils compiled from 61 studies in the conterminous United States showed lower bulk density and higher percent organic C and N than salt marshes, regardless of geographic region. Salinity, a proxy for freshwater input, was inversely correlated with percent soil organic C and N and with vertical accretion in Georgia marshes and in marshes elsewhere in the conterminous United States. There was no relationship between above- or belowground emergent plant production and salinity of Georgia marshes but the rate of root decomposition was positively related to salinity, and decomposition rate was negatively related to percent soil organic C and C accumulation. In Georgia tidal marshes and elsewhere, soil organic matter content and accumulation are mediated by freshwater through its effects on decomposition.

The mixing of freshwater and seawater is a fundamental driving force that determines the structure and function of tidal marshes. Freshwater governs the distribution of emergent plant communities and tidal marsh fauna along the estuarine gradient (Simpson et al. 1983; Odum 1988). Spatial patterns in biogeochemical processes such as sediment deposition (Paludan and Morris 1999), P sorption (Sundareshwar and Morris 1999), denitrification (Seitzinger 1988), methanogenesis (Bartlett et al. 1987), and sulfate reduction (Capone and Kiene 1988) also are linked to spatial and temporal variation in freshwater input.

Soils are an important component of tidal marshes. They sequester organic matter, N, and P (Craft et al. 1988), support complex biogeochemical reactions (Capone and Kiene 1988), and contribute to long-term marsh stability through deposition of mineral sediment and accumulation of organic matter (DeLaune et al. 1983; Hatton et al. 1983; Nyman et al. 1990; Craft et al. 1993; Morris et al. 2002). Some work to date has suggested that soil properties vary with freshwater input along the estuarine gradient. Along the Louisiana Gulf coast, percent organic C and N were greatest in tidal freshwater marshes and decreased along the salinity gradient to salt marshes (Hatton et al. 1983). Bulk density was lower in tidal freshwater marshes than in salt marshes (Hatton et al. 1983; Nyman et al. 1990). In the Cooper River estuary (South Carolina), tidal marshes exhibited similar patterns in bulk density and percent C, N, and P along the gradient from tidal freshwater marsh to salt marsh (Paludan and Morris 1999; Sundareshwar and Morris 1999).

Soil accretion, the change in vertical elevation, also is linked to freshwater input (Stevenson et al. 1986; Nyman et al. 1990; DeLaune et al. 2003). Riverine marshes along the Nantikoke River (Maryland) and Savannah River (Georgia) exhibited higher rates of accretion than nonriverine estuarine marshes though much of the difference was attributed to greater sediment input in riverine marshes (Stevenson et al. 1986). Along the Louisiana Gulf coast, vertical accretion was higher in brackish and tidal freshwater marshes than salt marshes (Hatton et al. 1983) or there was no clear trend in accretion among marshes along the salinity gradient (Nyman et al. 1990).

Results from the studies above suggest that tidal marsh soil properties vary along the gradient from tidal freshwater marshes to salt marshes. It is not clear though, whether these patterns are consistent across different geographic regions, nor is there a clear understanding of the processes that drive these patterns. To resolve these issues, I measured soil properties (bulk density; percent organic C, N, P), vertical accretion, and mass accumulation of sediment, organic C, N, and P in nine tidal marshes on the Georgia coast that vary in freshwater input. I

[1] Corresponding author (ccraft@indiana.edu).

Acknowledgments

The idea for this work grew out of two National Science Foundation (NSF)–sponsored workshops, *Regulation of Organic Matter Preservation in Wetland Sediments* held at NSF Long Term Ecological Research (LTER) All Scientists Meeting, Snowbird, Utah (1–5 August 2000) and *Soil Organic Matter in Wetlands* held at Virginia Institute of Marine Sciences, Gloucester Point VA (26–29 July 2001). Thanks to Ken Helm, Sean Graham, and Josh Hall for their assistance in the field; Chrissy Pruett, Jillian Bertram, and Sarah Butler for preparing and analyzing soil samples; and Steve Pennings, Dale Bishop, and Samantha Joye for sharing data. Steve Pennings, Pat Megonigal, Ross Brittain, and two anonymous reviewers provided constructive comments on previous versions of the manuscript.

This work was supported by grant OCE-9982133 from the National Science Foundation and is a contribution of the Georgia Coastal Ecosystems LTER program.

This is Contribution 889 of the University of Georgia Marine Institute.

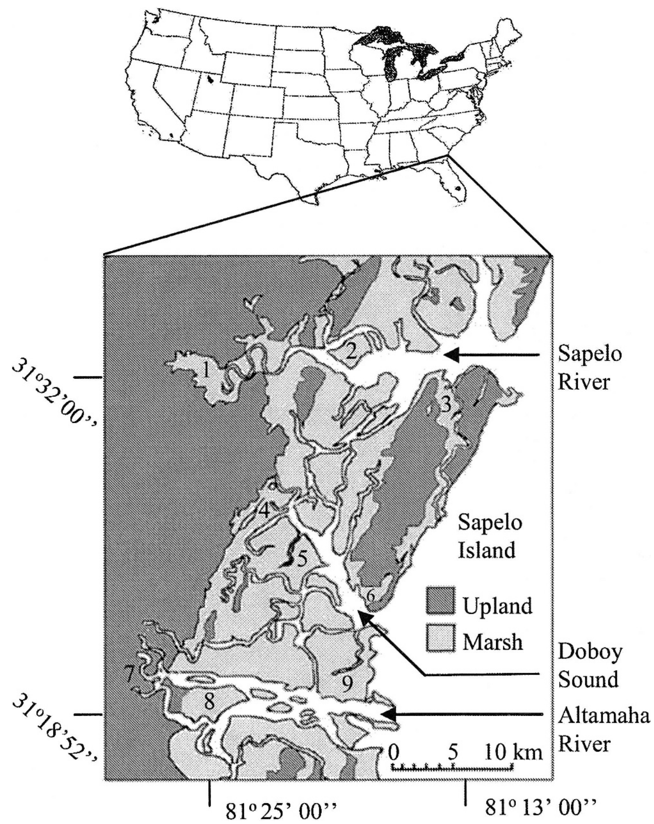


Fig. 1. Location of freshwater- (Altamaha River) and marine- (Doboy Sound, Sapelo River) dominated marshes along the Georgia coast. Numbers on the figure correspond to the stations in Table 1.

hypothesized that enhanced delivery of freshwater would lead to reduced bulk density and greater percent C, percent N, vertical accretion, and accumulation of soil organic matter and nutrients. I also measured root decomposition and productivity in four marshes along the salinity gradient to test the role of these processes in controlling soil organic C stocks and accumulation. In addition, I surveyed soil properties of tidal marshes of the conterminous United States (northeast and southeast Atlantic, Gulf, West coasts) to ascertain whether the patterns observed in Georgia marshes are representative of tidal marsh systems in other geographic regions.

Methods

Site description—Marshes of three estuaries on the Georgia coast, representing low (Sapelo River, Doboy Sound) and high (Altamaha River) inputs of freshwater, were selected for sampling (Fig. 1). Within each estuary, I sampled three marshes arrayed along an onshore–offshore gradient and consisting of upstream, intermediate, and downstream landscape positions (Table 1). I used salinity as a proxy for freshwater input. Along the Altamaha River—the freshwater-dominated estuary—mean surface water salinity (2002 and 2003) in the adjacent river ranged from 0.15 at the upstream location to 16.5 downstream (Table 1). Salinities at comparable locations in Doboy Sound and Sapelo River were substantially greater, ranging from 23 to 30, except for the landward location on Sapelo River (site 1, Eulonia), where salinity was 13.5. Tide range was similar at all sites, including those along the salinity gradient, and was approximately 2.3 m.

Vegetation of the Altamaha River marshes consisted of smooth cordgrass (*Spartina alterniflora* Loisel) at the downstream location, giant cordgrass (*Spartina cynosuroides* (L.); levee), and *Juncus roemerianus* Scheele (marsh plain) at the intermediate brackish marsh location, and giant cutgrass (*Zizaniopsis milaceae*) (Michaux) at the upstream tidal freshwater marsh location (Table 1). In Doboy Sound and Sapelo River, *S. alterniflora* was the dominant species at all landscape positions except for the upstream landscape position of the Sapelo River (site 1), where *J. roemerianus* was the dominant species on the marsh plain.

Soil sampling and lab analyses—In 2001, one soil core, 8.5 cm diameter by 30 cm deep, was collected from the streamside zone and one from the marsh plain at the nine sampling locations. Cores were sectioned into 2-cm depth

Table 1. Site number, name, estuary, dominant water source, surface water salinity, and dominant vegetation of the nine marsh study sites. See Fig. 1 for location of the sites. For each estuary, sites 1, 4, and 7 are farthest inland.

Site No.	Name	Estuary	Water source	Salinity*	Dominant vegetation
1	Eulonia	Sapelo River	Marine influence	12–15	*Spartina alterniflora*, *Juncus roemerianus*
2	Four-mile			25–29	*S. alterniflora*
3	N Sapelo			28–32	*S. alterniflora*
4	Meridian	Doboy Sound	Marine influence	20–25	*S. alterniflora*
5	Folly River			22–25†	*S. alterniflora*
6	Dean Creek			25–29	*S. alterniflora*
7	Carr's Island	Altamaha River	Freshwater influence	0–0.3	*Zizaniopsis milaceae*
8	Alligator Creek			2–6	*Spartina cynosuroides*, *J. roemerianus*
9	Rockdedundy Island			14–19	*S. alterniflora*, *J. roemerianus*

* Annual (2003 and 2002) salinity calculated from daily measurements in the water column of the adjacent estuary.
† Interpolated from measurements at the nearest locations, Meridian and Dean Creek (*see Fig. 1*).

increments in the field and transported to the lab where they were air-dried, weighed for bulk density, then ground and sieved through a 2-mm-diameter mesh screen. Ground soil was packed into 50-mm-diameter by 9-mm-deep petri dishes and analyzed for ^{137}Cs to determine vertical accretion on the basis of gamma analysis of the 661.62 keV photopeak (Craft et al. 2003).

Subsamples of ground soil were analyzed for organic C and total N with a Perkin-Elmer 2400 CHN analyzer. Samples were tested for presence of carbonates by adding a drop of dilute (0.1 mol L^{-1}) HCl then observing whether effervescence occurred. Samples containing carbonates were pretreated with 0.1 mol L^{-1} HCl before CHN analysis. Total P was determined by colorimetric analysis after digestion in nitric-perchloric acid (Sommers and Nelson 1972). Percent sand, silt, and clay (0–10 cm) was determined by the hydrometer method (Gee and Or 2002).

Short-term sediment accretion was measured with feldspar marker plots (Cahoon 1994). Three feldspar plots (0.25 by 0.25 m^2) were established on the marsh plain at each site in June 2003. Small-diameter soil cores (four per plot) were collected in January 2004 after 6.5 months. At some salt marsh sites (2, 3, 6), the feldspar marker layer could not be discerned because of bioturbation by fiddler crabs.

Root decomposition and in-growth—Root decomposition and productivity were measured in a subset of marshes (sites 6, 7, 8 , 9), in which salinity ranged from 0 to 27, following the methods of Blum (1993). Nylon mesh bags (2 by 2 mm), 30 cm long by 7 cm wide, were filled with 10 g of fresh roots collected from the dominant vegetation of the levee and plain of each marsh (Table 1). Sixteen bags containing native roots were deployed on each levee, and 16 were deployed on each marsh plain for a total of 128 bags. Bags were buried to a depth of between 10 and 20 cm in June 2003. Approximately every 3 months for 1 yr (364 d), four bags each were retrieved from the levee and the marsh plain at each site. Bags were dried at 70°C to a constant weight and weighed. Before deployment, a subset of four bags from each marsh location was dried immediately to determine the initial (time 0) dry mass. Soil temperature (n = 5 per marsh location) was measured seasonally at the same time that root bags were retrieved by inserting a temperature probe 10 cm into the soil.

Root production was determined by the in-growth method (Blum 1993). Every 3 months when bags were retrieved, new roots that grew into the bags were carefully removed by hand. Fresh roots were washed with distilled water to remove soil material and dried at 70°C to a constant weight. Root production was calculated as the sum of new roots that grew into each of the four bags collected seasonally during the 1-yr sampling period.

Geographic comparison of soil properties—Soil properties and accumulation of sediment, organic C, N, and P were compared in tidal freshwater, brackish, and salt marshes of the northeast Atlantic (New Jersey to Maine), southeast Atlantic (Florida to Delaware), Gulf coast (Texas to Florida), and West Coast (California to Washington). A total of 61 published studies and two unpublished datasets, including this one, were used in the statistical analysis (*see* Web Appendix 1, Table A1, http//www.aslo.org/lo/toc/vol_52/issue_3/1220a1.pdf). Forty published studies and this study were used in the analysis of vertical accretion and sediment, C, N, and P accumulation (*see* Web Appendix 1, Table A2).

Statistical analysis—Differences in soil properties, accretion, and accumulation among the three estuaries were tested by analysis of variance followed by post hoc comparisons using the Ryan-Einot-Gabriel-Welsch Multiple Range Test (REGWQ. SAS 1996). Differences between the marsh levee and plain were tested by Student's t-test. Statistical analysis of soil properties ($n = 18$ cores) was performed on mean values integrated over the 30-cm depth ($n = 15$ depth increments). Because ^{137}Cs profiles from four levee sites were uninterpretable, statistical analysis of accretion; sediment deposition; and C, N, and P accumulation were based on 14 rather than 18 cores.

Analysis of variance and REGWQ also were used to test for differences among tidal freshwater, brackish, and salt marshes of the conterminous United States and for differences among geographic regions for a given marsh type (i.e., salt marshes). I excluded Texas (salt) marshes from the statistical analysis because they differed so much from the Louisiana salt marshes that receive large amounts of freshwater and sediment relative to the Texas marshes. For statistical analyses, I defined tidal freshwater marshes as having salinity < 0.5, brackish marshes with salinity $=$ 0.5–15, and salt marshes with salinity > 15. The salinity ranges also corresponded to the dominant vegetation (e.g., *S. alterniflora* in salt marshes) for each marsh type.

Differences in root decomposition and production were determined with a two-way analysis of variance on the basis of marsh type and location (levee, streamside). Main effects means were separated by REGWQ. Decomposition was modeled by an exponential decay regression of dry mass remaining versus time. Correlation and linear regression were used to explore relationships among shoot and root production, root decomposition, environmental variables (salinity, temperature, root C : N, surface water inorganic N, P, and particulate C, crab burrows), percent organic C and C accumulation. For all statistical analyses, tests of significance were based on $\alpha = 0.05$.

Results and discussion

Vertical accretion—Fourteen of 18 soil cores contained interpretable ^{137}Cs profiles with well-defined peaks (Table 2), and four levee sites (2, 3, 5, 9) lacked distinct ^{137}Cs maxima in the soil profile. Marshes of the freshwater-dominated Altamaha River had higher rates of vertical accretion ($p < 0.05$) compared with marshes of the marine-dominated Doboy Sound and Sapelo River (Fig. 2a). Short-term accretion measured by feldspar marker layers also was significantly greater in the Altamaha River marshes than in the marshes of Doboy Sound (Fig. 2b), and it was positively correlated with ^{137}Cs-based measurements of accretion ($r = 0.85$, $p < 0.05$, $n = 6$ plain sites).

Table 2. Distribution of ^{137}Cs with depth and ^{137}Cs inventories (0–30 cm) in Sapelo River, Doboy Sound, and Altamaha River marsh soils. ^{137}Cs maximum (±1 counting error) of interpretable cores is shown in bold.

	^{137}Cs (Bq kg^{-1})					
	Site 1		Site 2		Site 3	
Depth (cm)	Levee	Plain	Levee	Plain	Levee	Plain
Sapelo River						
0–2	18.5	**8.14 ± 0.7**	0	1.85	4.44	6.66
2–4	**30.34 ± 1.7**	1.11	1.48	2.22	6.29	**7.77 ± 1**
4–6	24.42	0.00	1.85	2.59	5.55	7.77
6–8	22.94	1.11	1.48	2.96	5.92	6.66
8–10	16.28	0.00	2.59	3.70	5.92	5.92
10–12	16.28	0.37	4.81	2.96	6.29	3.70
12–14	11.10	1.48	3.70	2.59	4.44	4.07
14–16	4.81	0	3.33	**6.29 ± 1.0**	5.18	4.44
16–18	3.70	0.74	5.55	5.92	5.18	3.33
18–20	2.96	0	3.70	3.33	5.55	2.96
20–22	1.85	0.74	1.48	4.44	5.92	0
22–24	2.22	0	4.07	2.22	6.29	0
24–26	1.85	1.48	1.48	0.74	5.55	2.96
26–28	0	1.85	0	1.48	6.29	0
28–30	1.85	0.00	0	0	6.29	0
Inventory (Bq cm^{-2}):	80.7	30.7	31.1	37.0	59.2	46.3
	Site 4		Site 5		Site 6	
Doboy Sound						
0–2	18.5	**8.14 ± 1.0**	0.48	4.44	5.92	**14.1 ± 1**
2–4	**30.34 ± 1.0**	1.11	4.81	8.14	5.55	12.95
4–6	24.42	0	7.03	**11.84 ± 1.9**	**31.45 ± 0.9**	10.36
6–8	22.94	1.11	5.55	3.70	4.81	7.77
8–10	16.28	0	4.44	2.59	5.92	4.07
10–12	16.28	0.37	6.29	3.33	3.70	2.96
12–14	11.10	1.48	6.66	2.22	2.22	3.70
14–16	4.81	0	5.92	2.22	2.96	2.96
16–18	3.70	0.74	7.03	2.59	0	2.22
18–20	2.96	0	7.03	0	0	1.85
20–22	1.85	0.74	5.55	0.74	0.74	1.85
22–24	2.22	0	5.18	0	0.37	0
24–26	1.85	1.48	2.59	0	0.37	0
26–28	0	1.85	2.96	0	0	0
28–30	1.85	0	2.96	0	0	0
Inventory (Bq cm^{-2}):	38.5	13.3	43.3	30.0	47.0	51.1
	Site 7		Site 8		Site 9	
Altamaha River						
0–2	23.31	37.00	11.84	14.43	5.92	0
2–4	27.75	41.44	13.32	9.99	5.92	2.22
4–6	32.56	52.54	12.95	21.83	3.33	4.44
6–8	36.26	57.72	13.69	**26.64 ± 2.2**	5.92	**7.92 ± 1**
8–10	40.70	61.42	17.76	23.31	5.18	4.07
10–12	46.62	56.98	20.72	25.53	3.70	4.07
12–14	46.20	**100.27 ± 2.7**	21.09	22.57	4.44	3.33
14–16	50.32	73.63	29.97	22.94	4.00	4.81
16–18	**58.09 ± 2.1**	17.39	37.00	21.09	7.40	4.40
18–20	19.61	3.33	**46.62 ± 2.1**	16.65	5.18	4.29
20–22	2.22	2.22	37.37	7.03	4.81	4.07
22–24	0	0	22.20	5.55	5.55	3.70
24–26	1.11	1.11	17.39	2.96	5.55	3.70
26–28	1.11	0	12.95	0	5.92	0
28–30	0	1.11	9.25	2.59	5.92	0
Inventory (Bq cm^{-2}):	212.7	194.6	202.8	85.1	66.6	43.3

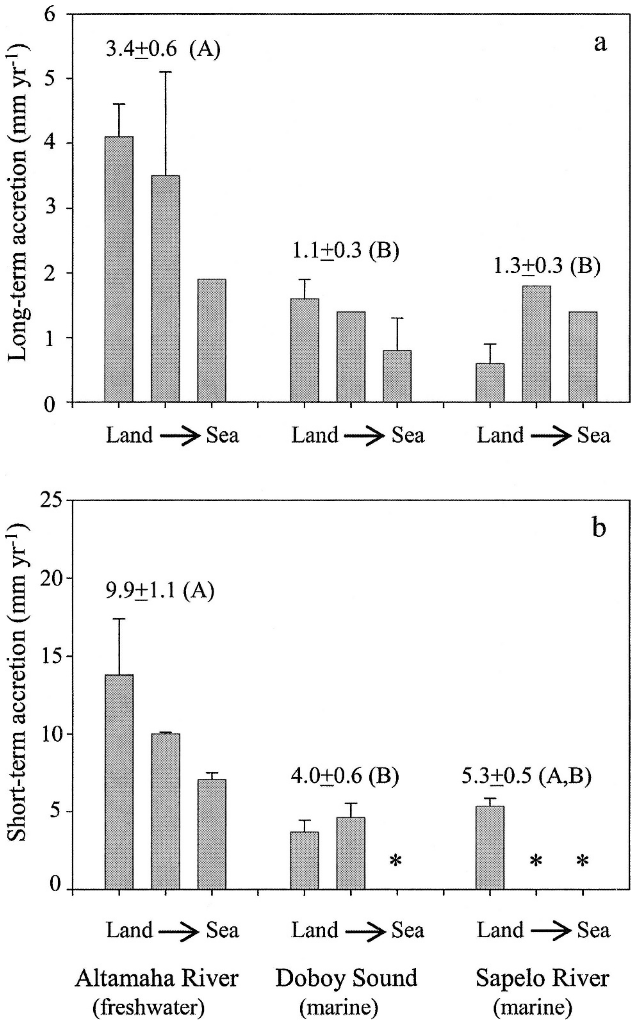


Fig. 2. (a) Long-term (^{137}Cs) and (b) short-term (feldspar marker) vertical accretion of freshwater- (Altamaha River) and marine- (Doboy Sound, Sapelo River) dominated marshes along the Georgia coast. Means $\pm$ 1 SE are shown for each estuary. Means separated by the same letter (in parentheses) are not significantly different ($p \leq 0.05$) on the basis of the Ryan-Einot-Gabriel-Welsch multiple range test. Locations marked with an asterisk (*) were omitted: the feldspar marker layer could not be detected because of bioturbation by fiddler crabs.

Sapelo River marshes exhibited statistically intermediate rates of short-term accretion (5.3 $\pm$ 0.5 mm yr^{-1}), but mean rates were close to those at Doboy Sound though bioturbation was problematic and reduced the sample size in these estuaries. Among individual marshes, short-term accretion was greatest in the tidal freshwater marsh (14 $\pm$ 4 mm yr^{-1}), the upstream location of the Altamaha River (site 7), compared with other marshes (4–10 mm yr^{-1}) (Fig. 2b).

Cs-137 is known to be fixed by micaceous clay minerals in soils and sediments but, in anaerobic lake sediments, can be remobilized by ion exchange with NH_4^+ (Comans et al. 1989). Remobilization of ^{137}Cs is greater initially (2–3% after year 1) but declines with time (0.5% per year after 30 yr) as it is buried by accumulating sediment (Smith and Comans 1996). After 30 yr, up to 20% of the ^{137}Cs inventory may be remobilized. In salt marshes, there is evidence for downward diffusion of ^{137}Cs relative to ^{241}Am, another artificial radionuclide marker that is less mobile than ^{137}Cs, but the ^{137}Cs peak remains in place (Thomson et al. 2002). Also, in salt marshes, it appears that the location of the ^{137}Cs peak is unaffected by differences in redox (oxidized vs. reduced) state (Thomson et al. 2002).

Evidence for ^{137}Cs remobilization in Georgia tidal marsh soils comes from ^{137}Cs inventories (0–30 cm depth) that range from 7% (site 4, plain) to 118% (site 7, levee) of decay-corrected atmospheric deposition (180 Bq cm^{-2}), with an overall mean of 73 Bq cm^{-2} or 41% of atmospheric deposition. ^{137}Cs inventories were greatest in tidal freshwater marshes and brackish marshes of the Altamaha River (Table 2) that had the highest percent soil organic C (Table 3) and that received greater inputs of terrestrially derived riverborne particles (Fig. 2b). The ^{137}Cs inventory was lowest at site 4 (plain), where percent organic matter was low (Table 3). The ^{137}Cs maximum was located at the soil surface (Table 2); hence, accretion was negligible. Similar to this study, ^{137}Cs inventories in marine sediments were greater in areas that receive ^{137}Cs associated with riverborne particles (Su and Huh 2002). And in tidal marsh soils, terrestrial soils, and marine sediments, ^{137}Cs inventories were positively related to organic matter content (McHenry and Ritchie 1977; Park et al. 2004). Regardless of ^{137}Cs remobilization that could result in low ^{137}Cs inventories in Georgia tidal marsh soils, the similarity in trends of ^{137}Cs- and feldspar-based accretion rates provides convincing evidence that freshwater input promotes vertical accretion in these marshes.

Soil organic C and N—Percent organic C and C accumulation were two and three times greater, respectively, in marshes of the freshwater-dominated Altamaha River than in marshes of Doboy Sound and Sapelo River (Table 3; Fig. 3a). Greater percent organic C and C accumulation in low-salinity marshes might be attributed to enhanced plant productivity (NPP) that adds organic C to the soil or suppressed decomposition that preserves C, or both. In situ measurements with buried roots suggest C accumulation is linked to freshwater input through its effect on decomposition. After 1 yr, mass loss of "native" roots, that is roots collected from emergent vegetation growing at each site (*see Table 1*), was significantly greater ($p < 0.001$) in two salt marshes (41–49%) than in the brackish marsh (29–30%) and the tidal freshwater marsh (30–36%). The rate of decomposition (k) was positively related to water column salinity (Fig. 4a) but not to litter quality (e.g., nutrients, lignin), which is known to affect decomposition rates (Melillo et al. 1982; Conn and Day 1997). Although litter quality differed among the source materials, there was no relationship between decomposition expressed as decay rate coefficients (k) and root C : N (r^2 = 0.01), one common measure of litter quality (C.B. Craft unpubl. data). Decomposition rate also was unrelated to surface water nutrient concentrations. Saline marshes exhibiting the highest rates of decomposition had lower concentrations of dissolved inorganic N (DIN, 3.8–6 μmol

Table 3. Soil bulk density, organic C, total N, total P, C : N, and N : P (0–30 cm depth) of freshwater-dominated (Altamaha River) versus marine-dominated marshes (Doboy Sound, Sapelo River).*

		Bulk density ($g\ cm^{-3}$)		Organic C (%)		Nitrogen (%)		Phosphorus ($\mu g\ g^{-1}$)		C : N (mol:mol)		N : P (mol:mol)	
Site No.	Estuary	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain	Levee	Plain
1	Sapelo River	0.23	1.09	11.3	1.4	0.51	0.08	450	260	26	20	25	7
2		0.47	0.43	4.1	4.2	0.23	0.25	580	670	21	20	9	8
3		0.39	0.41	5.1	4.2	0.35	0.30	860	590	17	16	9	11
4	Doboy Sound	1.10	1.03	1.4	2.1	0.09	0.13	100	80	17	19	20	35
5		0.30	0.35	5.5	6.6	0.37	0.36	450	600	18	21	18	13
6		0.41	0.41	5.2	4.6	0.34	0.32	520	520	18	17	15	13
7	Altamaha River	0.32	0.19	8.0	14.7	0.54	0.93	560	520	17	18	21	39
8		0.31	0.22	10.1	16.2	0.65	0.84	640	560	18	23	22	33
9		0.27	0.38	7.9	4.1	0.50	0.40	510	648	19	18	21	9
Mean Sapelo ($n = 6$)		0.50±0.12 a		5.0±1.4 a		0.29±0.06 a		570±80 a		20±1 a		11±3 a	
Mean Doboy ($n = 6$)		0.60±0.15 a		4.2±0.8 a		0.27±0.05 a		380±90 a		18±1 a		19±3 ab	
Mean Altamaha ($n = 6$)		0.28±0.03 a		10.2±1.8 b		0.64±0.08 b		570±20 a		18±1 a		25±4 b	

* Estuary means separated by the same letter are not significantly different ($\alpha<0.05$) according to the Ryan-Einot-Gabriel-Welsch multiple range test.

L^{-1}) and P (0.26–0.37 μmol L^{-1}) than the tidal freshwater and brackish marshes (DIN = 13.5–14 μmol L^{-1}, dissolved inorganic P = 0.38–0.49 μmol L^{-1}; Samantha Joye unpubl. data). The relationship between k and soil temperature (10 cm depth) , another factor that affects decomposition rate, was not significant ($r^2 = 0.35$, $p = 0.12$). Root decomposition has been shown to vary spatially in salt marshes, with greater decomposition at higher elevations on the levee (Hemminga et al. 1988). However, in Georgia marshes, there was no relationship between decomposition and levee versus plain locations (Fig. 4a).

The positive relationship between decomposition and salinity could be linked to the availability of sulfate for anaerobic decomposition or to biotic factors linked to salinity.

Sulfate, a terminal electron acceptor for anaerobic decomposition, is readily supplied by inundation with seawater, and in brackish marshes and salt marshes, sulfate reduction is the dominant pathway of anaerobic organic matter mineralization. In a laboratory study, salinity intrusion of 10 ppt to Georgia tidal freshwater sediments doubled anaerobic organic matter mineralization rates compared with sediments that were exposed to freshwater only. After 4 weeks of exposure, sulfate reduction accounted for 95% of total organic matter mineralization (Weston et al. 2006). These findings contrast with field-based studies that reported no difference in anaerobic organic matter mineralization in brackish versus tidal freshwater marsh soils (Neubauer et al. 2005) and sediments (Kelley et al. 1990).

Of unknown importance is the contribution of aerobic respiration to explaining the patterns of decomposition, percent soil organic C, and C accumulation observed in Georgia marshes. Bioturbation by fiddler crabs could promote aerobic decomposition through burrowing that promotes oxygen diffusion into the soil (Montague 1980; Bertness 1985; Kostka et al. 2002). In this study, the relationship between density of crab burrows and decomposition rate was significant, with greater decomposition in salt marshes where fiddler crab burrows were most abundant (Fig. 4b).

Whereas soil organic C content and accumulation were related to decomposition rate (Fig. 4c,d), there was no relationship between aboveground NPP (Steve Pennings unpubl. data) and percent organic C ($r^2 = 0.01$, $n = 18$) or organic C accumulation ($r^2 = 0.12$, $n = 14$). There also was no relationship between root production and percent C ($r^2 = 0.05$) or C accumulation ($r^2 = 0.09$; $n = 4$ marshes, levee and plain locations), suggesting that marsh soil organic matter dynamics are not determined by inputs of NPP.

Sediment deposition is another factor that could promote accumulation of soil organic matter (DeLaune et al. 2003) by burying it while diluting organic C content of the soil. In Georgia marshes, I observed greater C accumulation in tidal freshwater and brackish marshes, in which sediment deposition on feldspar marker layers was greater, than in salt marshes (Fig. 2b). However, percent organic C content also was greater in the tidal freshwater and brackish marshes, suggesting that burial by itself does not regulate organic matter dynamics of tidal marsh soils.

Finally, allochthonous organic matter deposited on the marsh surface during tidal flooding could conceivably explain the high organic C observed in Altamaha River marshes. However, there was no relationship between particulate carbon measured seasonally in nearby estuarine waters (June 2003–March 2004; Samantha Joye unpubl. data) and percent soil organic C ($r^2 = 0.12$, $p = 0.18$). Also, organic C accumulation in soil was negatively related to water column particulate carbon ($r^2 = 0.50$, $p < 0.01$). Thus, allochthonous C inputs to Georgia tidal marsh soils appear to be low to negligible, which is corroborated by studies from a variety of marsh types and geographic locations, indicating that tidal marsh soil organic matter is derived mostly from accumulation of belowground biomass produced by emergent vegetation (McCaffrey and Thomson 1980; Hatton et al. 1983; Bricker-Urso et al. 1989; Nyman et al. 1990, Blum 1993).

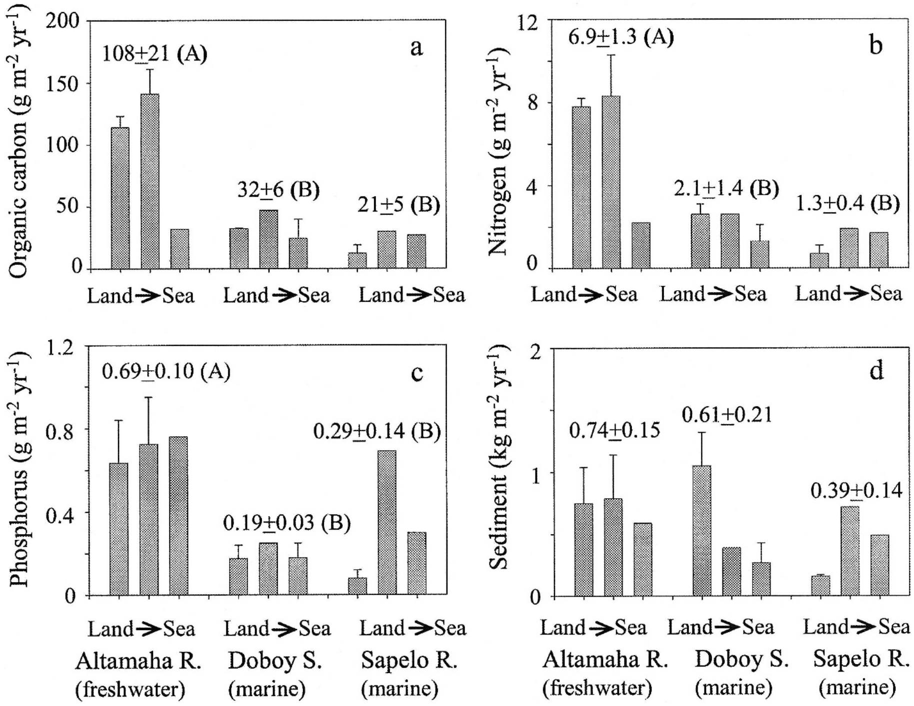


Fig. 3. ^{137}Cs-based measurements of (a) organic C, (b) N, (c) P, and (d) mineral sediment accumulation in freshwater- (Altamaha River) and marine- (Doboy Sound, Sapelo River) dominated marshes along the Georgia coast. Means ± 1 SE are shown for each estuary. Means separated by the same letter (in parentheses) are not significantly different ($p \leq 0.01$) based on the Ryan-Einot-Gabriel-Welsch Multiple Range Test.

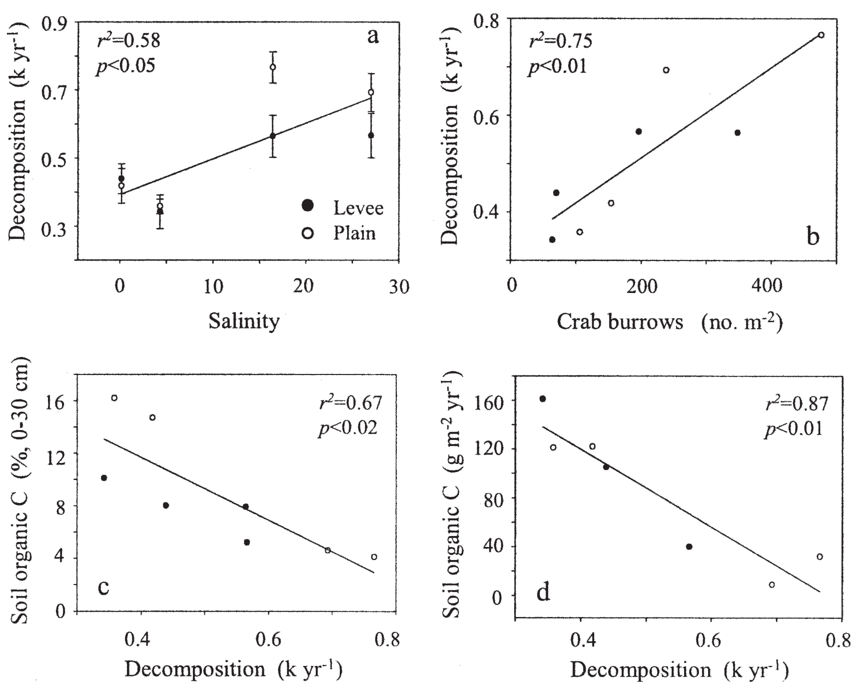


Fig. 4. Relationships between decomposition rate and (a) salinity and (b) crab burrow density, and (c) percent soil organic C and (d) organic C accumulation versus decomposition rate in four tidal marshes exposed to different salinities. Salinity is based on daily (2002 and 2003) measurements in the water column of the adjacent estuary. Burrow density is based on spring and fall 2003 counts of crab burrows (n = 4) per marsh zone per season (Dale Bishop unpubl. data).

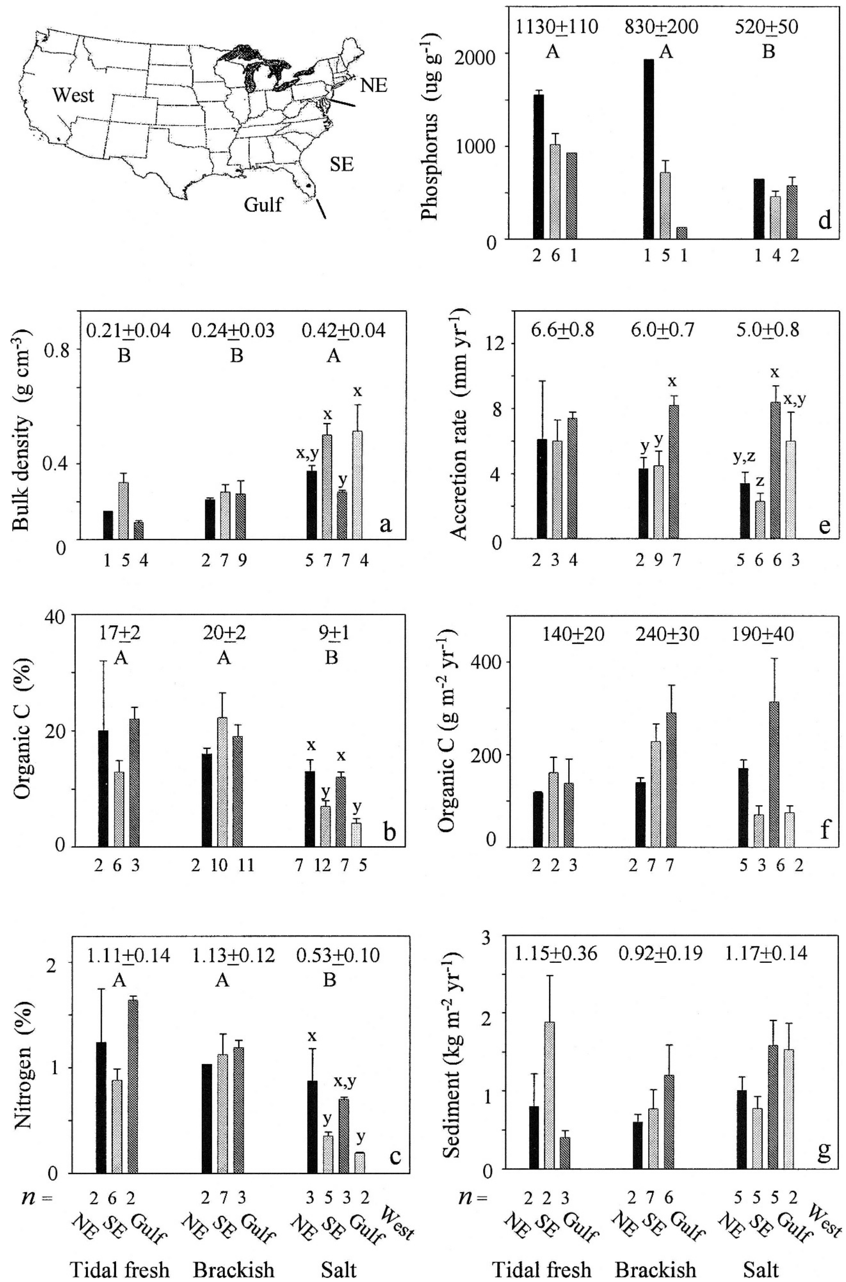


Fig. 5. Comparison of (a) bulk density, (b) percent organic C, (c) percent N, (d) P content, (e) accretion rate, (f) organic C accumulation, and (g) mineral sediment accumulation in northeast (NE) and southeast Atlantic, Gulf coast (excluding Texas salt marshes), and West coast tidal marshes of the continental United States. Bars and error bars represent the mean ± 1 SE. Means separated by the same letter (A, B for marsh type; x, y, z for region within marsh type) are not significantly different ($p < 0.05$) according to the Ryan-Einot-Gabriel-Welsch multiple range test). Data and references used to construct the graphs are presented in Web Appendix 1 (Tables A1, A2).

Soil N, which exists mostly (95%) in organic matter (Craft et al. 1991), and N accumulation also were two and three times greater, respectively, in freshwater-dominated marshes of the Altamaha River than in marshes of Doboy Sound and Sapelo River (Table 3; Fig. 3b). There was no difference in soil C : N among marshes of the three estuaries although N : P was significantly greater in marshes of the Altamaha River compared with marshes of Sapelo Sound (Table 3). Greater N : P in freshwater-dominated marshes was attributed to greater percent N rather than lesser amounts of P (Table 3).

Other soil properties—Bulk density did not differ among marshes of the three estuaries, although bulk density in the

Table 4. Correlations of soil properties with salinity for tidal marshes of Georgia and the conterminous United States.

	Georgia	Conterminous U.S.
Soil properties		
Bulk density (g cm^{-3})	0.25	0.63***
Organic C (%)	−0.70**	−0.53**
Organic C (mg cm^{-3})	−0.70**	−0.11
Nitrogen (%)	−0.73***	−0.49*
Phosphorus (μg g^{-1})	0.03	−0.49*
C : N (mol : mol)	−0.16	0.05
N : P (mol : mol)	−0.63	−0.07
Soil accumulation		
Feldspar accretion (mm yr^{-1})	−0.94**	—
^{137}Cs accretion (mm yr^{-1})	−0.69**	−0.47*
Sediment (g m^{-2} yr^{-1})	ns	−0.06
Organic C (g m^{-2} yr^{-1})	−0.79***	−0.27
Nitrogen (g m^{-2} yr^{-1})	−0.79***	−0.57*
Phosphorus (g m^{-2} yr^{-1})	−0.61**	−0.05

ns, Not significant.
* $p<0.05$.
** $p<0.01$.
*** $p<0.001$.

freshwater-dominated Altamaha River marshes was half that measured in the salt marshes (Table 3). There also was no difference in soil properties related to the mineral (nonorganic) fraction. For example, soil P (Table 3) and percent sand, silt, and clay did not differ between freshwater-dominated marshes and salt marshes (data not shown). In spite of greater vertical accretion in Altamaha River marshes (Fig. 2), there was no difference in long-term (^{137}Cs) sediment accumulation among marshes of the three estuaries (Fig. 3d). Phosphorus accumulation, however, was two to three times greater in freshwater-dominated Altamaha River marshes than marine-dominated marshes (Fig. 3c) and was attributed to greater vertical accretion in these marshes.

Comparison with tidal marsh soils of the conterminous United States—As found in Georgia marshes, tidal fresh and brackish marsh soils across all geographic regions had significantly lower bulk density and greater percent organic C and N than salt marshes (Fig. 5a–c). Tidal fresh and brackish marsh soils also contained more P than salt marshes (Fig. 5d), which contrasts with my findings of comparable P concentrations among Georgia marshes.

In marshes of the conterminous United States, vertical accretion, although not significantly different among marsh types, exhibited the same trend as observed in Georgia marshes: decreasing rate of accretion with increasing salinity (Fig. 5e). Organic C and N accumulation and N : P were greater in tidal fresh- and brackish-water marshes than in salt marshes of Georgia (Fig. 3; Table 3), but I did not see the same trends when comparing tidal marshes across geographic regions (Fig. 5f). There was no difference in sediment deposition among tidal marshes of Georgia or among marshes of different geographic regions (Fig. 5g), nor were differences noted in soil C : N (range 17–20), N : P (23–24), N accumulation (9–15 g m^{-2} yr^{-1}), or P accumulation (0.8–1.3 g m^{-2} yr^{-1}) among tidal freshwater, brackish, and salt marshes of different geographic regions.

Salt marshes were present in the four geographic regions, which allowed for comparison of soil properties among the NE and SE Atlantic, Gulf (Louisiana), and West coasts. Bulk density of Louisiana Gulf Coast salt marshes was half (0.26 g cm^{-3}), and organic C (12%) and N (0.70%) were double, those of salt marshes of the SE Atlantic and West coasts (Fig. 5a–c). Mississippi River flooding also enhanced vertical accretion of Louisiana salt and brackish marshes relative to comparable marshes in other regions (Fig. 5e). Although not significantly different among regions, organic C accumulation (Fig. 5f) also trended higher in Louisiana salt and brackish water marshes. These findings suggest that organic matter is more important to vertical accretion and long-term stability of subsidence-prone Louisiana marshes relative to marshes in other geographic regions.

Differences in soil properties also were evident between Louisiana marshes and other Gulf coast marshes. Brackish and salt marshes of the Louisiana coast contained more organic C than comparable marshes in Texas, Mississippi, and Florida. Percent organic C of Louisiana salt marshes (12 ± 1%; n = 7) was greater than in Texas salt marshes (4 ± 2%; n = 2). Also, brackish marshes of Louisiana contained more organic C (16 ± 2%; n = 7) than brackish marshes of Mississippi and Florida (9 ± 2%, n = 4).

Salt marsh soils of the SE Atlantic coast were more similar to West coast marshes than to NE Atlantic and Gulf coast salt marshes. Bulk density, organic C, and N were comparable in salt marshes of the two regions (Fig. 5–c), both of which have relatively high evapotranspiration and salinity. Salt marshes of the NE Atlantic coast have lower bulk density and high C and N relative to SE Atlantic coast salt marshes (Fig. 5a–c). These marshes are exposed to a cooler climate that probably slows decomposition and preserves soil organic C and N relative to SE Atlantic salt marshes.

In Georgia marshes and in marshes of the conterminous United States, percent soil organic C and N, vertical accretion, and N accumulation were negatively correlated with salinity (Table 4). Soil N : P and C and P accumulation also were negatively correlated with salinity in Georgia tidal marshes, but not in marshes of the conterminous United States. In U.S. tidal marshes, bulk density was positively correlated with salinity, and P concentration was negatively correlated with salinity but I did not see the same trends in Georgia marshes.

In Georgia tidal marshes and elsewhere, freshwater input promotes organic matter preservation and accumulation. In Georgia, short- and long-term accretion, percent soil organic C, N and N : P, and accumulation of organic C and N were greater in tidal marshes of the freshwater-dominated Altamaha River than in salt marshes of Doboy Sound and Sapelo River. In situ decomposition of roots was greater in salt marshes than in the tidal freshwater and brackish marshes and it was positively related to surface water salinity. Percent soil organic C and organic C accumulation were inversely related to decomposition but were unrelated to above- or belowground emergent plant production. Freshwater-driven, landscape-scale patterns of soil properties observed in Georgia tidal marshes also occur in other geographic regions of the conterminous United States. In a survey of 61 published and two unpublished studies, bulk density was lower and percent organic C, N, and P were consistently greater in tidal freshwater marshes and brackish marshes than in salt marshes regardless of geographic region. These findings suggest that freshwater input is important in structuring tidal marsh soils across a wide range of climatic and geomorphic conditions because of its association with lower decomposition rates relative to areas with greater seawater influence.

References

BARTLETT, K. B., D. S. BARTLETT, R. C. HARRISS, AND D. I. SEBACHER. 1987. Methane emissions along a salt marsh salinity gradient. Biogeochemistry **4:** 183–202.

BERTNESS, M. D. 1985. Fiddler crab regulation of *Spartina alterniflora* production on a New England salt marsh. Ecology **66:** 1042–1055.

BLUM, L. K. 1993. *Spartina alterniflora* root dynamics in a Virginia marsh. Mar. Ecol. Prog. Ser. **102:** 169–178.

BRICKER-URSO, S., S. W. NIXON, J. K. COCHRAN, D. J. HIRSCHBERG, AND C. HUNT. 1989. Accretion rates and sediment accumulation in Rhode Island salt marshes. Estuaries **12:** 300–317.

CAHOON, D. R. 1994. Recent accretion in two managed marsh impoundments in coastal Louisiana. Ecol. Appl. **4:** 166–176.

CAPONE, D. G., AND R. P. KIENE. 1988. Comparative microbial dynamics in marine and freshwater sediments: Contrasts in anaerobic carbon catabolism. Limnol. Oceanogr. **33:** 725–749.

COMANS, R. N. J., J. J. MIDDLEBURG, J. ZONDERHUIS, J. R. W. WOITTIEZ, G. J. DE LANGE, H. A. DAS, AND C. H. VAN DER WEIJDEN. 1989. Mobilization of radiocaesium in pore water of lake sediments. Nature **339:** 367–369.

CONN, C. E., AND F. P. DAY, JR. 1997. Root decomposition across a barrier island chronosequence: Litter quality and environmental controls. Plant Soil **195:** 351–365.

CRAFT, C. B., S. W. BROOME, AND E. D. SENECA. 1988. Nitrogen, phosphorus and organic carbon pools in natural and transplanted marsh soils. Estuaries **11:** 272–280.

———, ———, AND ———. 1991. Porewater chemistry of natural and created marsh soils. J. Exp. Mar. Biol. Ecol. **152:** 187–200.

———, E. D. SENECA, AND S. W. BROOME. 1993. Vertical accretion in regularly and irregularly flooded microtidal estuarine marshes. Estuar. Coast. Shelf Sci. **37:** 371–386.

———, AND OTHERS. 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. Ecol. Appl. **13:** 1417–1432.

DELAUNE, R. D., R. H. BAUMANN, AND J. G. GOSSELINK. 1983. Relationships among vertical accretion, coastal submergence and erosion in a Louisiana gulf coast marsh. J. Sediment. Petrol. **53:** 147–157.

———, A. JUGSUJINDI, G. W. PETERSON, AND W. H. PATRICK, JR. 2003. Impact of Mississippi River freshwater reintroduction on enhancing marsh accretionary processes in a Louisiana Estuary. Estuar. Coast. Shelf Sci. **58:** 653–662.

GEE, G. W., AND D. OR. 2002. Particle-size analysis, p. 255–293. *In* J. H. Dane and G. C. Topp [eds.], Methods of soil analysis, part 4: Physical methods. Soil Science Society of America.

HATTON, R. S., R. D. DELAUNE, AND W. H. PATRICK, JR. 1983. Sedimentation, accretion and subsidence in marshes of Barataria Basin, Louisiana. Limnol. Oceanogr. **25:** 494–502.

HEMMINGA, M. A., C. J. KOK, AND W. DE MUNCK. 1988. Decomposition of *Spartina anglica* roots and rhizomes in a salt marsh of the Westerschelde estuary. Mar. Ecol. Prog. Ser. **48:** 175–184.

KELLEY, C. A., C. S. MARTENS, AND J. P. CHANTON. 1990. Variations in sedimentary carbon remineralization rates in the White Oak River estuary, North Carolina. Limnol. Oceanogr. **35:** 373–383.

KOSTKA, J. E., B. GRIBSHOLT, E. PETRIE, D. DALTON, H. SKELTON, AND E. KRISTENSEN. 2002. The rates and pathways of carbon oxidation in bioturbated salt marsh sediments. Limnol. Oceanogr. **47:** 230–240.

MCCAFFREY, R. J., AND J. THOMSON. 1981. A record of the accumulation of sediment and trace metals in a Connecticut salt marsh. Adv. Geophys. **22:** 165–236.

MCHENRY, J. R., AND J. C. RITCHIE. 1977. Physical and chemical parameters affecting transport of ^{137}Cs in arid watersheds. Water Resour. Res. **13:** 923–927.

MELILLO, J. M., J. D. ABER, AND J. F. MURATORE. 1982. Nitrogen and lignin control of hardwood leaf litter decomposition dynamics. Ecology **63:** 621–626.

MONTAGUE, C. L. 1980. The influence of fiddler crab burrows and burrowing on metabolic processes in salt marsh sediments, p. 283–301. *In* V. S. Kennedy [ed.], Estuarine comparisons. Academic Press.

MORRIS, J. T., P. V. SUNDARESHWAR, C. T. NIETCH, B. KJERFVE, AND D. R. CAHOON. 2002. Responses of coastal wetlands to rising sea level. Ecol. **83:** 2869–2877.

NEUBAUER, S. C., K. GIVLER, S. K. VALENTINE, AND J. P. MEGONIGAL. 2005. Seasonal patterns and plant mediated controls of subsurface wetland biogeochemistry. Ecology **86:** 3334–3344.

NYMAN, J. A., R. D. DELAUNE, AND W. H. PATRICK, JR. 1990. Wetland soil formation in the rapidly subsiding Mississippi River deltaic plain: Mineral and organic matter relationships. Estuar. Coast. Shelf Sci. **31:** 57–69.

ODUM, W. E. 1988. Comparative ecology of tidal freshwater and salt marshes. Ann. Rev. Ecol. Syst. **19:** 147–176.

PALUDAN, C., AND J. T. MORRIS. 1999. Distribution and speciation of phosphorus along a salinity gradient in intertidal marsh sediments. Biogeochemistry **45:** 197–221.

PARK, G., AND OTHERS. 2004. Properties of ^{137}Cs in marine sediments off Yangnam, Korea. J. Environ. Radioact. **77:** 285–299.

SAS. 1996. SAS User's Guide. Statistical Analysis Systems Institute.

SEITZINGER, S. P. 1988. Denitrification in freshwater and coastal marine ecosystems: Ecological and geochemical significance. Limnol. Oceanogr. **33:** 702–724.

SIMPSON, R. L., R. E. GOOD, M. A. LECK, AND D. F. WHIGHAM. 1983. The ecology of freshwater tidal wetlands. BioScience **33:** 255–259.

SMITH, J. T., AND R. N. J. COMANS. 1996. Modeling the diffusive transport and remobilisation of ^{137}Cs in sediments: The effects of sorption kinetics and reversibility. Geochim. Cosmochim. Acta **60:** 995–1004.

SOMMERS, L. E., AND D. W. NELSON. 1972. Determination of total phosphorus in soils: A rapid perchloric acid digestion procedure. Soil Sci. Soc. Am. Proc. **36:** 902–904.

STEVENSON, J. C., L. G. WARD, AND M. S. KEARNEY. 1986. Vertical accretion in marshes with varying rates of sea level rise, p. 241–259. *In* D. A. Wolfe [ed.], Estuarine variability. Academic Press.

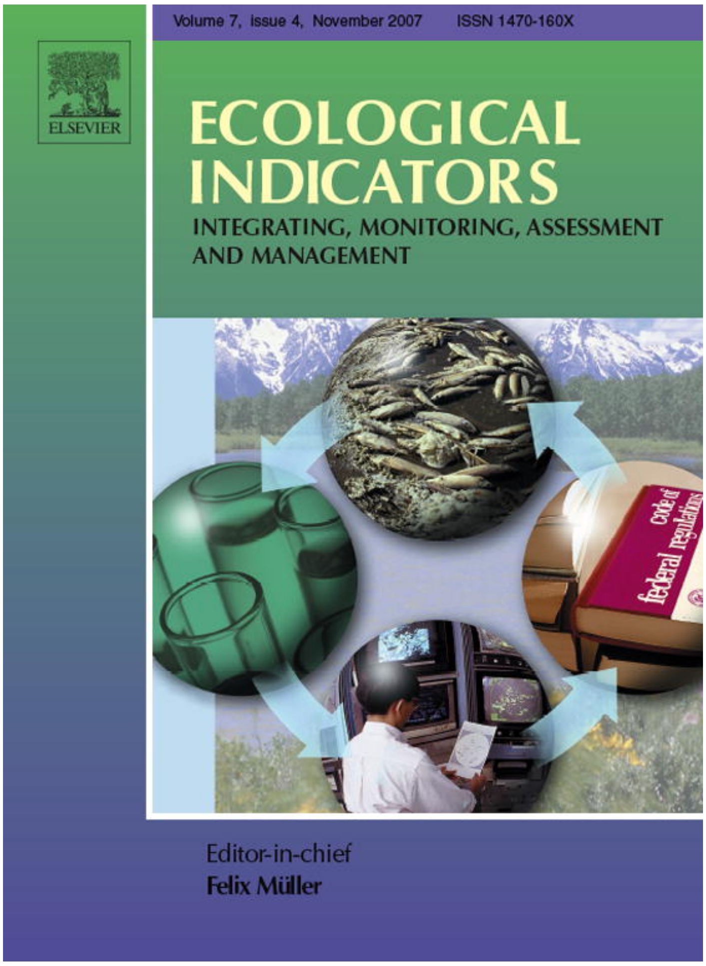
SU, C., AND C. HUH. 2002. ^{210}Pb, ^{137}Cs and ^{239,240}Pu in East China Sea sediments: Sources, pathways and budgets of sediments and radionuclides. Mar. Geol. **183:** 163–178.

SUNDARESHWAR, P. V., AND J. T. MORRIS. 1999. Phosphorus sorption characteristics of intertidal marsh sediments along an estuarine salinity gradient. Limnol. Oceanogr. **44:** 1693–1701.

THOMSON, J., F. M. DYER, AND I. W. CROUDACE. 2002. Records of radionuclide deposition in two salt marshes in the United Kingdom with contrasting redox and accumulation conditions. Geochim. Cosmochim. Acta **66:** 1011–1023.

WESTON, N. B., R. E. DIXON, AND S. B. JOYE. 2006. Ramifications of salinity intrusion in tidal freshwater sediments: Geochemistry and microbial pathways of organic matter mineralization. J. Geophys. Res, Biogeosci. **111,** G01009, doi:10.1029/2005JG000071.

Received: 19 June 2006
Accepted: 4 December 2006
Amended: 28 December 2006


Volume 7, Issue 4, November 2007
ISSN 1470-160X
ELSEVIER
ECOLOGICAL INDICATORS
INTEGRATING, MONITORING, ASSESSMENT AND MANAGEMENT
code of federal regulations
Editor-in-chief
Felix Müller

ELSEVIER

Ecological Indicators 7 (2007) 733–750

ECOLOGICAL INDICATORS

This article is also available online at:
www.elsevier.com/locate/ecolind

Ecological indicators of nutrient enrichment, freshwater wetlands, Midwestern United States (U.S.)

Christopher Craft *, Kandice Krull, Sean Graham

School of Public and Environmental Affairs, Indiana University, Bloomington, IN 47405, United States

Received 28 March 2006; received in revised form 7 August 2006; accepted 12 August 2006

Abstract

Vegetation and soil indicators of nutrient condition were evaluated in 30 wetlands, 10 each in 3 Nutrient Ecoregions (NE) (*VI-Corn Belt and Northern Great Plains*, *VII-Mostly Glaciated Dairy Region*, *IX-Temperate Forested Plains and Hills*) of the Midwestern United States (U.S.) to identify robust indicators for assessment of wetland nutrient enrichment and eutrophication. Nutrient condition was characterized by surface water inorganic N (NH_4-N, NO_3-N) and P (PO_4-P) concentrations measured seasonally for 1 year, plant available and total soil N and P, and aboveground biomass, leaf N and P and species composition of emergent vegetation measured at the end of the growing season. Aboveground biomass, nutrient uptake and species composition were positively related to surface water NH_4-N (N) but not to PO_4-P or NO_3-N. Aboveground biomass and biomass of aggressive species, *Typha* spp. plus *Phalaris arundinacea*, increased asymptotically with surface water N whereas leaf P, senesced leaf N and senesced leaf P increased linearly with N. And, species richness declined with surface water N. Soil total P was positively related to surface water PO_4-P but it was the only soil indicator related to wetland nutrient condition. Individual regressions for each NE generally were superior to a single regression for all NEs. In NE VI (*Corn Belt*), few indicators were related to surface water N because of the high degree of anthropogenic disturbance (85% of the landscape is cleared) as compared to NEs VII and IX (24–53% cleared). Of the indicators evaluated, stem height ($r^2 = 0.42$ for all NEs, $r^2 = 0.56$ for NE VII + IX) and percent biomass of aggressive species, *Typha* spp. plus *Phalaris*, ($r^2 = 0.46$ for all NEs, $r^2 = 0.54$ for NE VII + IX), were the best predictors of wetland nutrient enrichment. Vegetation-based indicators are a promising tool for assessment of wetland nutrient condition but they may not be effective in NEs where landscape disturbance is intense and widespread.

Keywords: Eutrophication; Water quality; Nitrogen; Phosphorus; Standards

1. Introduction

Nutrient enrichment is an increasing threat to aquatic and wetland ecosystems. The best documented example of wetland eutrophication in North America is the Florida Everglades where, near canals that convey N

* Corresponding author. Tel.: +1 812 855 5971; fax: +1 812 855 7802.
E-mail address: ccraft@indiana.edu (C. Craft).

1470-160X/$ – see front matter
doi:10.1016/j.ecolind.2006.08.004

and P enriched agricultural drainage stimulates P uptake and growth of emergent vegetation (Davis, 1991; Miao and Sklar, 1998), periphyton (McCormick et al., 1996) and microbial activity (Qualls and Richardson, 2000; Wright and Reddy, 2001). Soil P pools, including porewater bulk soil, are enriched relative to areas distant from the source of nutrient loading (Koch and Reddy, 1992; DeBusk et al., 1994, 2001; Qualls and Richardson, 1995). And, in P enriched areas, native sawgrass (*Cladium jamaicense*) and slough vegetation are replaced by near-monoculture stands of cattail (*Typha domingensis*) (Jensen et al., 1995; Craft and Richardson, 1997).

Like the Everglades, wetlands in other regions exhibit a similar response to nutrient enrichment. Wetland vegetation responds to nutrient dosing by increasing nutrient uptake and biomass production (Aerts and Berendse, 1988; Hayati and Proctor, 1991; Verhoeven and Schmitz, 1991; Shaver and Chapin, 1995; Bridgham et al., 1996; Shaver et al., 1998, 2001) though the response depends on whether N, P or other nutrients are limiting. A decline in plant species richness often is seen with progressive enrichment (Vermeer, 1986; Pegtel et al., 1996; Shaver et al., 2001; Gustafson and Wang, 2002) as aggressive species such as *Typha* spp., *Phalaris arundinacea* and *Phragmites australis* invade and dominate the site (Chambers et al., 1999; Galatowitsch et al., 1999; Svengsouk and Mitsch, 2001; Green and Galatowitsch, 2002; Maurer and Zedler, 2002; Woo and Zedler, 2002). Except for the Everglades, though, where tens of millions of dollars have been spent to identify the causes, effects, indicators and thresholds of P enrichment, there has been little systematic effort to identify the origins (N or P), document the effects and identify indicators of wetland nutrient enrichment in other geographic regions.

We measured surface water nutrients (N, P) and vegetation-, litter- and soils-based indicators of nutrient enrichment in 30 freshwater wetlands spanning three Nutrient Ecoregions (NEs) in the Midwestern United States to answer the following questions: (1) Do predictive relationships exist between surface water nutrient concentrations and vegetation-, litter-, or soil-based indicators, and, if so, which indicators exhibit the strongest response? (2) Is the response linked to N or P? and (3) Are the relationships robust, that is, are they applicable across NEs or do they vary among NEs? We chose these indicators because they respond similarly to nutrient enrichment in a variety of wetland types and have been suggested as potential indicators of wetland enrichment (U.S. EPA, 2002).

The Nutrient Ecoregion approach, the classification of landscape units based on geology, physiography, vegetation, climate, soils, land use and other factors was derived from Omernick (1987) by the U.S. Environmental protection Agency to develop water quality (N, P) standards for rivers, streams and lakes of the U.S. (http://www.epa.gov/waterscience/criteria/nutrient). This approach takes into account differences in environmental factors that result in differences in water quality. For example, in NE VI (*Corn Belt*) which is underlain by fertile soils and is intensively farmed, rivers and streams contain more N (2.2 mg/L) and P (76 ug/L) than rivers and streams of NE VII (*Mostly Glaciated Dairy Region*) (TN = 0.5 mg/L, TP = 33 ug/L) and NE IX (*Southeastern Temperate Forested Plains and Hills*) (TN = 0.69 mg/L, TP = 37 ug/L) where the soils are less fertile and forest cover is greater. By stratifying sample collection among NEs, we partition some of the geographic variability in wetland nutrient condition, recognizing that some NEs (e.g. NE VI) will exhibit higher baseline levels of nutrients, and so ecological indicators may respond somewhat differently to nutrients than NEs with lower baseline concentrations of N and P. One goal of this study is to identify ecological indicators that can be used as nutrient (N, P) standards for wetlands, comparable to the ecologically based nutrient standards such as chlorophyll *a* that are used for water quality assessments of rivers, streams and lakes (http://www.epa.gov/waterscience/criteria/nutrient).

2. Materials and methods

2.1. Site description

Thirty freshwater wetlands, 10 from each NE (NE VI, *Corn Belt and Northern Great Plains*, NE VII, *Mostly Glaciated Dairy Region*, and NE IX, *Southeastern Temperate Forested Plains and Hills*) were selected for sampling (Fig. 1). Nutrient Ecoregions were derived from Omernick (1987) who classified broad landscape units based on geology, physiography, vegetation, climate, soils, land use and other

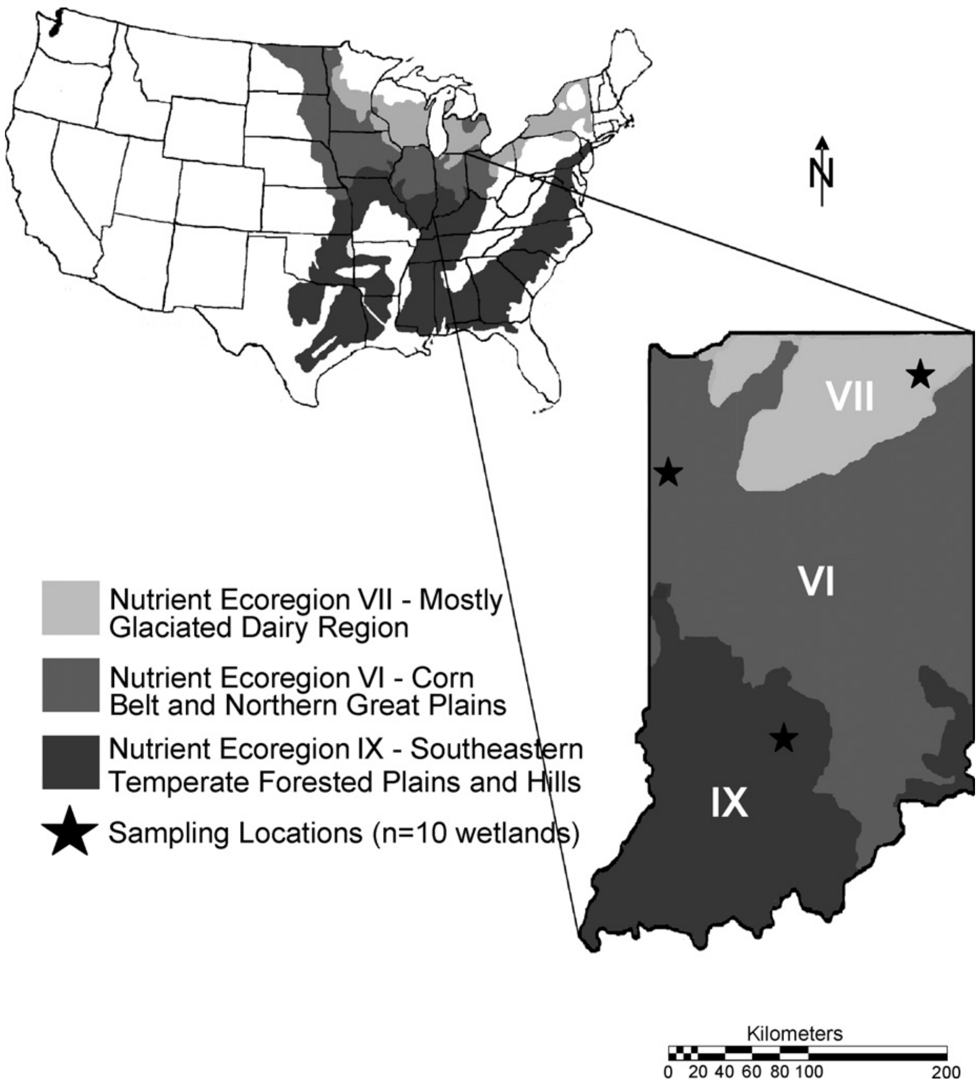


Fig. 1. Map of study area showing the three Nutrient Ecoregions (NE) and approximate sampling locations within each NE.

factors that determine nutrient concentrations in surface waters (http://www.epa.gov/waterscience/criteria/nutrient). Sandy-textured Mollisols and Entisols were common in NE VI (sandy, mixed, mesic Typic Haplaquolls and mixed, mesic Aquic Udipsamments) (USDA, 1998). Soils of NE VII consisted mainly of Mollisols (fine-loamy, mixed, mesic Typic Argiaquolls) and Histosols—organic soils (euic, mesic Typic Medisaprists) (USDA, 1981a,b). Nutrient Ecoregion IX was hillier than NEs VI and VII. It also had finer (silty-clayey) textured soils that were mostly Entisols (loamy-skeletal, mixed, acid, mesic Typic Udifluvents and fine-silty, mixed, acid, mesic Aeric Fluvaquents) (USDA, 1981a,b).

Within each NE, wetlands were chosen to encompass a range of nutrient enrichment. Wetlands containing high surface water nutrients were located in agricultural and urban catchments and included wetlands receiving treated wastewater from municipal wastewater treatment facilities. Low nutrient wetlands were situated in catchments that were mostly forested or open prairie and were located within nature preserves, protected areas and state owned wildlife management areas. To minimize differences in wetland age that may affect plant species composition (e.g. presence of pioneer species in young wetlands), we used topographic maps from the 1950s and 1960s to select wetlands that were at least 40–50 years old when we sampled in 2003.

2.2. Sample collection and analysis

2.2.1. Surface water N and P

Surface water inorganic N (NH_4-N, NO_3-N) and P (PO_4-P) were measured seasonally over 1 year (2003). On each sampling date, three water samples were collected from each wetland. Samples were filtered through 0.45 μm filter paper in the field and transported to the lab on ice. Ammonium-N and NO_3-N were determined by the phenate and cadmium reduction methods, respectively (APHA, 1998). Phosphate-P was determined using the ascorbic acid method (APHA, 1998).

2.2.2. Vegetation

Vegetation was sampled by clipping aboveground biomass from ten 0.25 m^2 quadrats in each wetland at the end of the growing season. Sampling was stratified to sample the two to three dominant zones in each wetland and, within each zone, samples were randomly collected. Height of the five tallest stems of the two to three dominant species was measured in the field. Plant material was transported to the lab where it was separated by species, then by live versus dead biomass. Vegetation was classified taxonomically based on Fassett (1957), Gleason and Cronquist (1991), and Voss (1996). Species lists for each site are presented in Appendices A–C.

Clipped material was dried at 70 °C to a constant weight. Because some species senesced earlier in the growing season (e.g. *Schoenoplectus* spp.) than others (e.g. *Typha*), aboveground live and standing dead biomass were combined for statistical analysis. Green and senesced leaves from the two to three dominant species in each quadrat were ground using a Wiley mill and analyzed for N, P and organic C. Nitrogen and organic C were measured using a Perkin-Elmer 2400 CHN analyzer. Total P was determined using the ascorbic acid method (APHA, 1998) after digestion in nitric-perchloric acid (Sommers and Nelson, 1972).

2.2.3. Litter and soil

Litter was collected from each 0.25 m^2 quadrat, dried at 70 °C to a constant weight, ground with a Wiley mill and analyzed for organic C, N and P. One soil core was collected from each quadrat using an 8.5 cm diameter by 10 cm deep stainless steel corer. Bulk density was measured by drying the soil at 70 °C, then dividing the dry mass by the volume of the corer. Plant available NH_4-N was extracted from field-moist soils with 2N KCl and analyzed using the phenate method (Mulvaney, 1996). Plant available PO_4-P of field-moist soils was extracted with sodium bicarbonate (Kuo, 1996) and analyzed by the ascorbic acid method (APHA, 1998). Total organic C and N of litter and soil were measured using a Perkin-Elmer 2400 CHN analyzer. Soils containing carbonates were pretreated with 0.1N HCl prior to CHN analysis to remove inorganic C. Total P was determined by the ascorbic acid method after digestion in nitric-perchloric acid (Sommers and Nelson, 1972). Soils data were expressed on a dry mass (g) and volume (cm^3) basis after correcting for water content determined by drying a 1 g field moist subsample at 105 °C.

2.3. Statistical analysis

Analysis of variance (ANOVA) was employed to test for differences in surface water nutrients, vegetation, litter and soils among the three NEs (SAS, 2002). Because no one plant species was present in all wetlands, leaf nutrient (N, P) concentrations for the two to three dominant species from a given site were pooled for statistical analysis. Aboveground biomass of aggressive species, reed canarygrass (*P. arundinacea*) and cattail (*Typha* spp.), also were combined for statistical analysis because the two species seldom were present together in the same wetland but it was common to find wetlands that were dominated by one species or the other. Means were separated using the Ryan-Einot-Gabriel-Welsch multiple range test (SAS, 2002). All test of significance were made at $\alpha = 0.05$.

Correlation analysis was used to investigate associations between surface water nutrients, vegetation indicators and soil indicators of nutrient enrichment. Regression analysis was used to explore relationships between surface water nutrient concentrations and vegetation, litter, and soil indicators for all NEs and for individual NEs. Linear, quadratic and asymptotic curve were evaluated and the best fit model was selected based on the maximum r^2 obtained (SYSTAT Software, 2004). Canonical correlation analysis (CCA) was used to identify trends of species abundance with wetland nutrient condition, surface water and soil nutrients (SAS, 2002). We dropped one site (MOR in NE VI) from the correlation, regression and CCA analyses because of its high NH_4-N (1.5 mg/L) and PO_4-P (0.92 mg/L) concentrations relative to other wetlands made it an influential data point (Rawlings et al., 2001). It should be noted that this site was dominated by a near monoculture of *Typha* (see Appendix A).

3. Results

3.1. Comparisons among NEs

Surface water inorganic N and P did not differ among the three NEs though NH_4-N and PO_4-P were somewhat greater in NE VI, *Corn Belt*, than in the other NEs (Table 1). In NE VI, one wetland (MOR) had high NH_4-N (1.5 mg/L) and PO_4-P (0.92 mg/L) relative to all other wetlands sampled (Table 1). If MOR is excluded, wetlands of NE VI still contain the most surface water NH_4-N (0.11 mg/L) though PO_4-P declines to 0.02 mg/L. Wetlands of NE VII exhibited high surface water NO_3-N relative to NEs VI and IX (Table 1) that was attributed to two fens, where average NO_3-N was 7.60 mg/L and 11.10 mg/L, respectively. Fens are groundwater-fed and the high NO_3-N concentrations likely are due to deep leaching of fertilizer nitrogen from agricultural fields in the region that discharges into these wetlands (Amon et al., 2002). In NE IX, the groundwater-fed Leonard Springs wetland also exhibited high NO_3-N (1.67 mg/L) as compared to other wetlands in this NE (0.01–1.11 mg/L).

Table 1
Nutrient-related properties of surface waters, vegetation and soils of freshwater wetlands of three Nutrient Ecoregions (NEs) of the Midwestern U.S.

	NE VI (*Corn Belt*)	NE VII (*Dairy Region*)	NE IX (*Forested Hills*)
Surface water			
NH_4-N (ug/L)	0.25 ± 0.14 (0.11 ± 0.03)[a]	0.09 ± 0.03	0.07 ± 0.01
NO_3-N (ug/L)	0.03 ± 0.01	2.0 ± 1.2	0.36 ± 0.18
PO_4-P (ug/L)	0.11 ± 0.09 (0.02 ± 0.01)[a]	0.06 ± 0.02	0.07 ± 0.04
Vegetation			
Stem height (cm)	161 ± 7 a	131 ± 6 b	134 ± 7 b
Aboveground biomass (g/m^2)	780 ± 90	630 ±60	890 ± 100
Species richness (#/site)	5.2 ± 0.9 a	10.1 ± 1.4 b	8.8 ± 0.9 b
Leaf P (ug/g)	1480 ± 60 a	1210 ± 60 b	1140 ± 70 b
Senesced leaf P (ug/g)	750 ± 60 a	530 ± 50 b	780 ± 50 a
Leaf N (%)	1.3 ± 0.1 b	1.5 ± 0.1 a	0.8 ± 0.04 c
Senesced leaf N (%)	0.8 ± 0.05	0.8 ± 0.04	0.7 ± 0.03
Leaf N:P (mol)	21.2 ± 1.1 b	34.8 ± 1.3 a	18.5 ± 0.8 b
Senesced leaf N:P (mol)	30.3 ± 4.5 b	48.4 ± 3.8 a	21.9 ± 1.0 b
Aggressive species (g/m^2)[b]	620 ± 100	400 ± 60	700 ± 140
Aggressive species (%)[b]	62 ±6	50 ± 5	49 ± 6
Litter			
Dry mass (g/m^2)	190 ± 50	131 ± 17	125 ± 20
Litter P (ug/g)	840 ± 100 b	570 ± 50 c	1080 ± 80 a
Litter N (%)	1.25 ± 0.28	0.87 ± 0.12	1.19 ± 0.11
Litter N:P (mol)	39 ± 3 a,b	48 ± 4 a	28 ± 2 b
Soils			
Available P (ug/cm^3)	9.3 ± 2 b	0.9 ± 0.1 c	13.5 ± 2.2 a
Total P (ug/cm^3)	170 ± 13 b	340 ± 20 b	410 ± 20 a
Available NH_4-N (ug/cm^3)	3.5 ± 0.5 b	5.2 ± 0.4 a	3.4 ± 0.3 b
Total N (mg/cm^3)	2.1 ± 0.2 b	4.2 ± 0.2 a	2.2 ± 0.1 b
Organic C (mg/cm^3)	28 ± 2 b	58 ± 2 a	27 ± 1.4 b

Means ($n = 10$) plus/minus one standard error are presented. Means separated by the same letter are not significantly different ($p = 0.05$) according to the Ryan-Einot-Gabriel-Welsch multiple range test.

[a] Minus site MOR.

[b] *Typha* spp. plus *Phalaris arundinacea*.

Stem height was greater and species richness was less in NE VI (*Corn Belt*) than in NEs VII and IX (Table 1). Aboveground biomass did not differ among NEs, ranging from 630 g/m^2 (NE VII) to 890 g/m^2 (NE IX). Nutrient Ecoregion IX, the southernmost NE, had the greatest biomass, perhaps due to the longer growing season, and it also contained the most biomass (700 g/m^2) of aggressive species. Proportionally though, biomass of aggressive species was greater in NE VI where they accounted for 62% of total biomass as compared to 49–50% for NE IX and VII (Table 1).

Leaf P was significantly greater in NE VI, where greater surface water N but not P concentrations were measured, than in the NE IX and NE VII (Table 1). And, senesced leaf P was greater in NEs VI and IX than in NE VII. Leaf N, C:N (data not shown) and N:P did not vary consistently with surface water N among the three NEs (Table 1). There also were no clear trends in litter mass, N or P among NEs that could be ascribed to differences in nutrient condition (Table 1).

Plant available P and total P were greater in NE IX than in NEs VI and VII (Table 1) and this difference was attributed to the fine textured (clayey) soils of NE IX (USDA, 1981a,b), that have high P sorption and, hence, high available and total P relative to the sandy soils of NE VI (USDA, 1998) and the organic soils of NE VII (USDA, 1981a,b). Nutrient Ecoregion VII, which had the lowest plant available P, also had the highest leaf, senesced and litter N:P of the three NEs (Table 1). Plant available NH_4-N and total N were significantly greater in NE VII that was attributed to the high organic carbon content of the soils of NE VII (Table 1). Across all NEs, soil total N ($r = 0.89$) and available N ($r = 0.55$) were positively correlated with organic C on a volume basis.

Correlation analysis revealed that vegetation indicators were more strongly associated with surface water nutrients, in particular NH_4-N, than with soil nutrients. Aboveground biomass ($r = 0.46$, $p < 0.01$), stem height ($r = 0.41$, $p < 0.05$), species richness ($r = -0.40$, $p < 0.05$) and biomass of aggressive species, *Typha* plus *Phalaris*, ($r = 0.46$, $p < 0.05$) were correlated with surface water NH_4-N but not PO_4-P or NO_3-N or with soil N or P. Leaf P ($r = 0.46$, $p < 0.01$), senesced leaf P ($r = 0.57$, $p < 0.01$) and senesced leaf N ($r = 0.48$, $p < 0.01$) also were positively correlated with surface water N. And, senesced leaf P was positively correlated with surface water PO_4-P ($r = 0.43$, $p < 0.05$) as well as soil extractable P on a mass ($r = 0.47$, $p < 0.01$) and a volume basis ($r = 0.46$, $p < 0.05$).

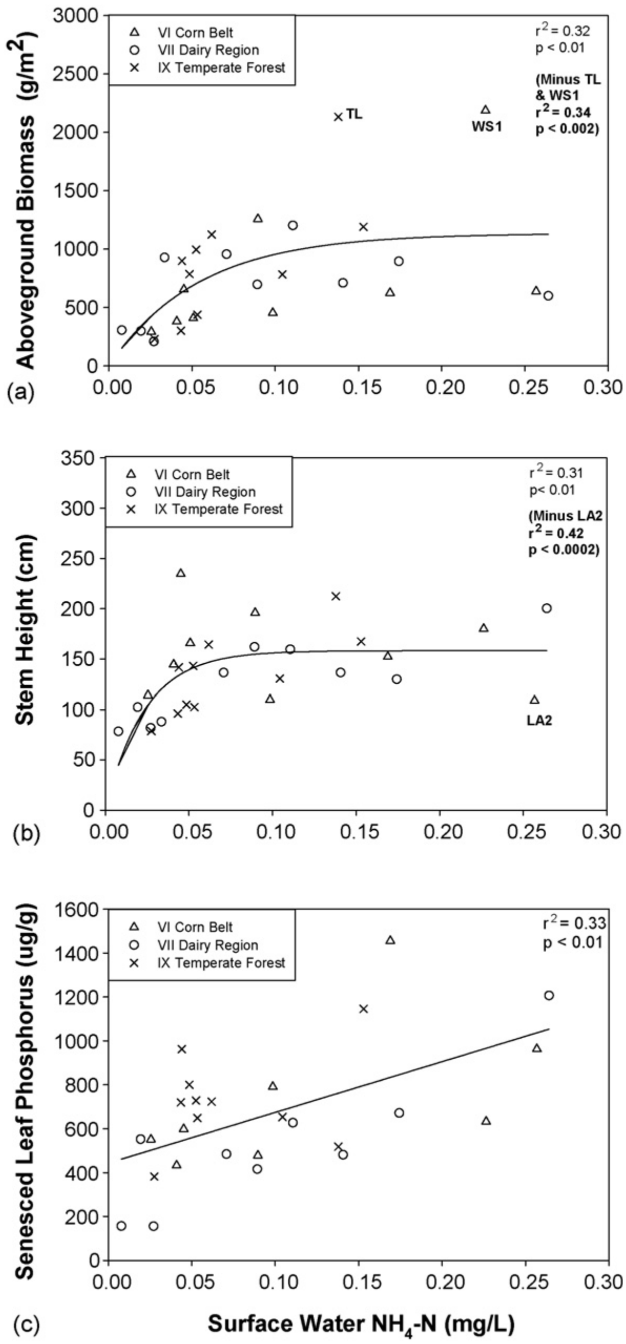


Fig. 2. (a) Aboveground biomass, (b) stem height and (c) senesced leaf P vs. mean surface water NH_4-N of 29 wetlands of NEs VI, VII and IX.

3.2. Indicators of nutrient enrichment: all NEs

Aboveground biomass and stem height increased asymptotically with NH_4-N (Figs. 2a and b). Senesced leaf P (Fig. 2c), leaf P ($r^2 = 0.21$, $p < 0.05$) and senesced leaf N ($r^2 = 0.24$, $p < 0.01$) also increased with surface water N. One would expect leaf and senesced leaf P to be related to surface water P rather than N though, in our wetlands, surface water NH_4-N and PO_4-P were positively correlated with each other ($r = 0.90$, $p < 0.0001$; minus site MOR, $r = 0.42$, $p < 0.05$). Species richness declined with surface water NH_4-N (Fig. 3a) while aggressive species, *Typha* spp. plus *P. arundinacea*, increased with surface water NH_4-N. Aboveground biomass of *Phalaris* plus *Typha* increased asymptotically with surface water NH_4-N (Fig. 3b). Of the vegetation indicators surveyed, percent biomass of aggressive species exhibited the strongest relationship with surface water N ($r^2 = 0.46$, $p < 0.0001$) (Fig. 3c).

Litter and soil indicators were not strongly related to wetland nutrient condition relative to vegetation. Litter P increased asymptotically with surface water NH_4-N ($r^2 = 0.26$, $p < 0.05$) but not with PO_4-P and soil total P (ug/g) increased asymptotically with surface water P ($r^2 = 0.26$, $p < 0.01$). There were no relationships between soil available P, available N and total N, and surface water nutrients.

3.3. Indicators of nutrient enrichment: individual NEs

We observed significant relationships between ecological indicators and surface water nutrients for NEs VII and IX but generally not for NE VI, *Corn Belt*. In NEs VII and IX, aboveground biomass increased with surface water NH_4-N and an asymptotic curve best fit the data for both NEs (Fig. 4a). Stem height also increased asymptotically and species richness declined linearly with surface water NH_4-N in the two NEs (Fig. 4b and c). Biomass of aggressive species (*Typha* plus *Phalaris*) increased asymptotically with surface water NH_4-N in NE VII and linearly with NH_4-N in NEs VI and IX (Fig. 5a). When expressed as the percentage of total biomass, aggressive species increased with surface water N in NEs VII and IX but not in NE VI (Fig. 5b). Green leaf P and senesced leaf P were positively related to surface water NH_4-N in NE VII but not in the other NEs (Table 2). Except for NE VII where litter P which was positively related to surface water NH_4-N (Table 2), we observed no relationships between litter and soil nutrients and surface water N and P for individual NEs.

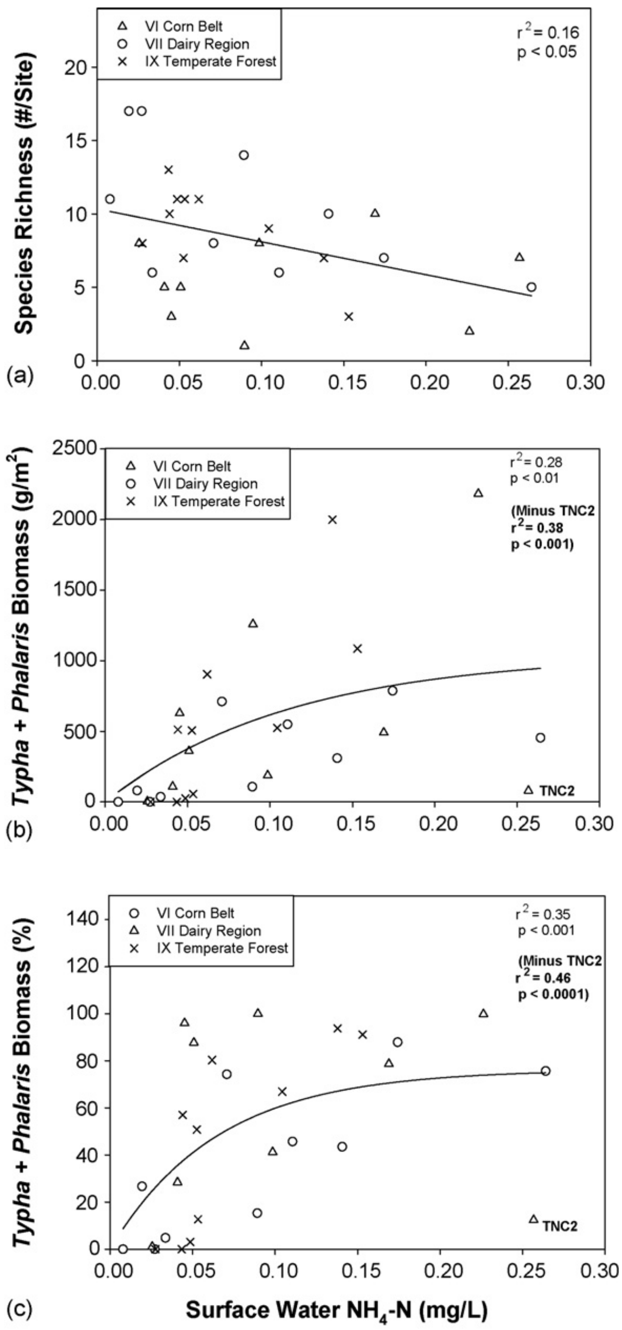


Fig. 3. (a) Species richness and biomass of aggressive species expressed as (b) g/m^2 and (c) percent of total community biomass vs. surface water NH_4-N of 29 wetlands of NEs VI, VII and IX.

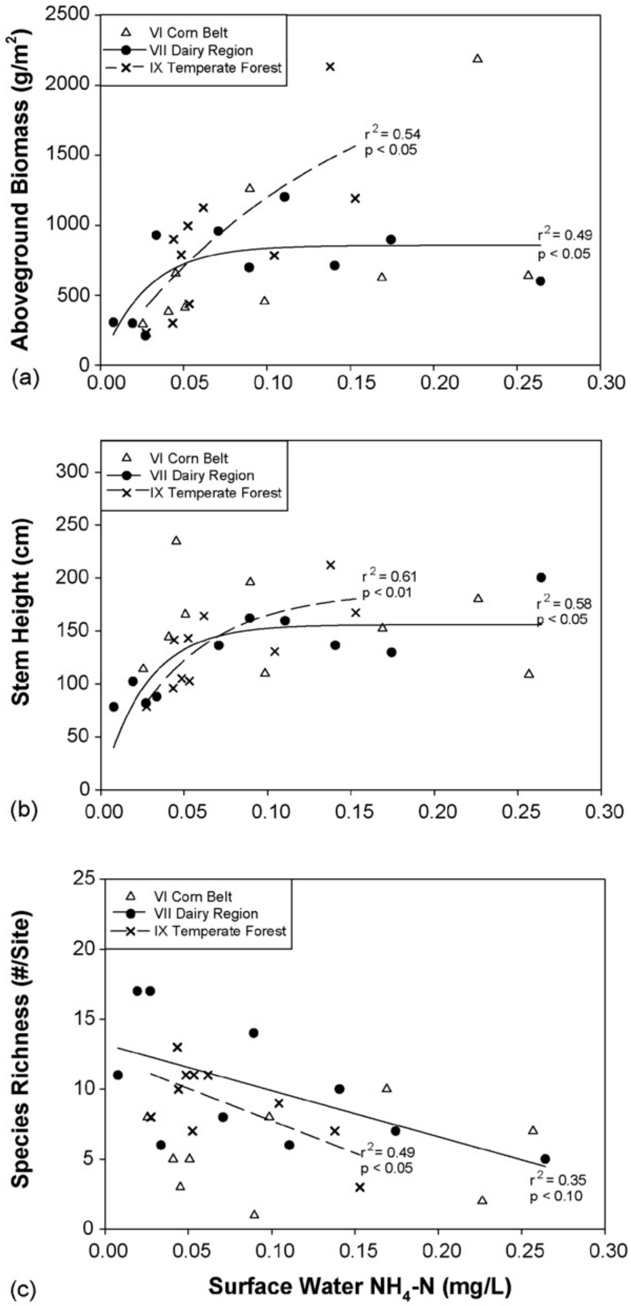


Fig. 4. Ecoregion-specific regressions of (a) aboveground biomass, (b) stem height and (c) species richness vs. surface water NH_4-N from 29 wetlands of NEs VII and IX. No significant relationships were observed for NE VI.

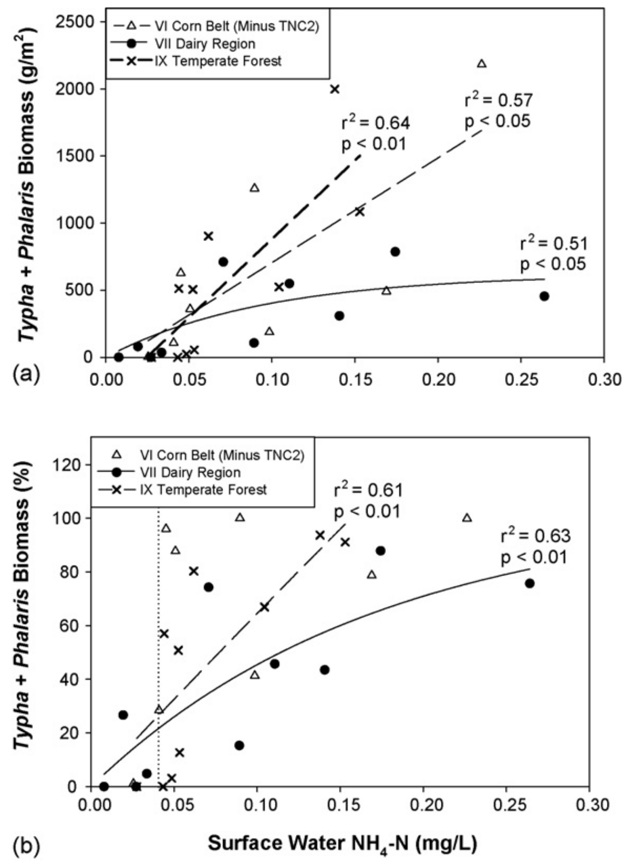


Fig. 5. Ecoregion-specific regressions of aboveground biomass of aggressive species expressed as (a) g/m^2 and (b) percent of total community biomass vs. surface water NH_4-N for 28 wetlands of NEs VI, VII and IX. Site TNC2 (NE VI) was dropped from the analysis.

4. Discussion

Nutrient enrichment leads to predictable changes in wetland structure and function, including increased N and P uptake and NPP (Davis, 1991; Miao and Sklar, 1998) and dominance by aggressive species (Jensen et al., 1995; Craft and Richardson, 1997; Chambers et al., 1999; Svengsouk and Mitsch, 2001; Green and Galatowitsch, 2002; Maurer and Zedler, 2002; Woo and Zedler, 2002) that leads to a decline in species richness (Vermeer, 1986; Drexler and Bedford, 2002; Gustafson and Wang, 2002). We report similar alteration of vegetation structure and function correlated with increasing surface water NH_4-N (Figs. 2 and 3) but not PO_4-P or NO_3-N. For example, across all NEs, indicators associated with NPP (aboveground biomass, stem height), nutrient uptake (leaf N, P) and dominance by aggressive species (*Typha*, *Phalaris*) increased with surface water NH_4-N and species richness declined (Table 2). Indicators of NPP (aboveground biomass, height, biomass of aggressive species) increased asymptotically with N, suggesting that the subsidy effect of increased N is

Table 2
Goodness of fit (r^2) of statistically significant relationships ($p < 0.05$) between ecological indicators and wetland nutrient condition

Indicator	Nutrient	Model	Nutrient ecoregion (NE)				
			All NEs	VI	VII	IX	VII + IX
Vegetation							
Aboveground biomass (g/m^2)	NH_4-N	Asymptotic	0.32	ns	0.49	0.54	0.36
Stem height (cm)	NH_4-N	Asymptotic	0.42[a]	ns	0.58	0.61	0.56
Green leaf P (ug/g)	NH_4-N	Linear	0.21	ns	0.43	ns	–[b]
Senesced leaf P (ug/g)	NH_4-N	Linear	0.33	ns	0.77	ns	–[b]
Senesced leaf N (%)	NH_4-N	Linear	0.24	ns	ns	ns	–[b]
Species richness (#/site)	NH_4-N	Linear	0.16	ns	0.35	0.49	0.33
Aggressive species (g/m^2)	NH_4-N	Linear	0.38[c]	0.57[c]	0.51	0.64[d]	0.58[e]
Aggressive species (%)	NH_4-N	Asymptotic	0.46[c]	ns	0.63	0.61	0.54
Litter							
Phosphorus (ug/g)	NH_4-N	Asymptotic	0.26	ns	0.67	ns	–[b]
Soil							
Total P (ug/cm^3)	PO_4-P	Asymptotic	0.26[f]	ns	ns	ns	–[b]

ns, not significant.
[a] Minus site LA2.
[b] Not analyzed.
[c] Minus site TNC2.
[d] Asymptotic fit.
[e] NE VI + IX.
[f] Minus site BOT.

diminished at higher surface water NH_4-N concentrations (Gerloff and Krombholz, 1966; Odum et al., 1979) whereas indicators of nutrient uptake (leaf N, green and senesced leaf P) and species richness exhibited a linear response to N. Our findings are consistent with results from the Florida Everglades where aboveground biomass, nutrient (P) uptake and dominance by *T. domingensis* are positively correlated with P enrichment (Koch and Reddy, 1992; Craft and Richardson, 1993, 1997; Miao and Sklar, 1998; Doren et al., 1999). Drexler and Bedford (2002) report a similar response (e.g. increased stem height and dominance by *Typha latifolia*, reduced species richness) to nutrient enriched agricultural drainage for a fen wetland in upstate New York.

We observed no relationships between ecological indicators and surface water NO_3-N which is not surprising since, in the saturated soils of wetlands, most inorganic N is in reduced form as ammonium (Ponnamperuma, 1972; Craft et al., 1991). However, other studies have demonstrated that emergent vegetation responds positively to nitrate. Addition of 0, 12 and 48 g NO_3-N/m^2 year as calcium nitrate to mesocosms stimulated growth in sedge meadow vegetation (11 species) of the presence and absence of *Phalaris* (Green and Galatowitsch, 2002). In mesocosms containing *Phalaris*, biomass of reed canary grass more than doubled in the high NO_3-N treatment (1426 g/m^2) relative to the control mesocosms (619 g/m^2).

Our regressions suggest that N as NH_4-N may limit or co-limit productivity in Midwestern wetlands. Studies from the region and elsewhere also suggest that emergent vegetation is limited by N or co-limited by N and P. In Europe, addition of N (but not P) or potassium (K) increased aboveground biomass of swale, fen and wet grasslands (Willis, 1963; Vermeer, 1986). In a fertilization study where N, P and/or K were added to 45 wetlands, N additions increased aboveground biomass in more cases (19) than any other nutrient or nutrient treatment (Verhoeven et al., 1996). Svengsouk and Mitsch (2001) added N, P and N + P to mesocosms containing *Schoenoplectus tabernaemontani* (aka *Scirpus validus*) and *T. latifolia* in Ohio. After 1 year, *Typha* produced significantly more aboveground biomass in the N + P treatment and, after 2 years, both species exhibited greater growth but only in response to N + P. Evidence to support nutrient co-limitation also comes from Wisconsin, where *Typha* × *glauca* grew more in N + P treated plots than in control plots but there was no response to N or P applied singly (Woo and Zedler, 2002).

Additional support for N limitation of Midwestern wetland vegetation comes from N:P ratios where, based on the threshold N:P of 33:1 (green leaves) suggested by Koerselman and Mueleman (1996) and Verhoeven et al. (1996), our results indicate that wetland vegetation in NEs VI and IX (N:P = 21) may be N limited. For NE VII, leaf N:P ratios of 35 and low available P in soil (Table 1) suggest co-limitation by N and P. Many western European wetlands are thought to be N limited based on N:P ratios (15–33) of fen and wet meadow vegetation (Venterink et al., 2002). In Ohio and Wisconsin, N:P ratios for *T. latifolia* (30) and *Typha* × *glauca* (<31) suggest N limitation or co-limitation by N and P (Svengsouk and Mitsch, 2001; Woo and Zedler, 2002). Some studies though caution against using biomass N:P as an index of N versus P limitation because, while it is sensitive to P limitation, it does not work as well for evaluating N limitation (Gusewell et al., 2003).

In our wetlands, species richness declined with surface water NH_4-N but the relationship was not strong (Fig. 3a). Increasing dominance by aggressive species, *Typha* and *Phalaris*, though was strongly related to surface water N (Fig. 3b and c). Reduced plant species diversity and increasing dominance by a few aggressive species has been reported in connection with nutrient enrichment of wetlands, including bogs, fens, wet meadows, riparian areas, and swamps (Vermeer, 1986; Galatowitsch et al., 1999; Drexler and Bedford, 2002; Gustafson and Wang, 2002; Childers et al., 2003). Nutrient enrichment enables aggressive species to out compete native species for light and space (Maurer and Zedler, 2002). *Typha* × *glauca*, a hybrid of *T. latifolia* and *T. angustifolia* and reed canary grass (*Phalaris arundincea*) are common invasive plants of Midwestern wetlands (Galatowitsch et al., 1999) and, in controlled experiments, both species respond positively to nutrient enrichment. In Wisconsin, additions of N + P (7:1 ratio) to greenhouse-grown *Typha* × *glauca* increased biomass production while, in a field experiment, additions of fertilizer (N, P, K) stimulated growth of *Typha* × *glauca* more than sedge meadow graminoids (Woo and Zedler, 2002). Similarly, in mesocosm and field experiments, addition of nutrients (N, P, K) promotes biomass production and dominance by *Phalaris* over other emergent species (Maurer and Zedler, 2002; Kercher and Zedler, 2004b). Our regressions suggest that, in Midwestern wetlands, expansion of *Typha* and *Phalaris*, is positively linked to nutrient enrichment, especially NH_4. It is important though to recognize that other anthropogenic disturbances (flooding, sediment, light) interact with nutrients to promote invasion by *Phalaris* and *Typha* as has been shown in greenhouse (Wetzel and van der Valk, 1998), mesocosm (Newman et al., 1996; Kercher and Zedler, 2004a,b) and field experiments (Maurer and Zedler, 2002).

In addition to the environmental factors mentioned above, wetland age and stage of succession also structures plant community composition. According to Clements (1928), succession proceeds in a predictable orderly manner as pioneer species, possibly *Typha* and *Phalaris* in wetlands, colonize the site and, over time, are replaced by succeeding assemblages of plants and leading to a stable, climax community. It is unlikely, however, that our wetlands represent the early stages of succession since they were selected using topographic maps from the 1950s and 1960s, more than 40 years prior to sampling them. Gleason (1927) suggested that succession is driven by stochastic events, fortuitous seed dispersal together with changing environmental conditions at the site that determine community composition over time. This model has been employed to describe succession in temperate freshwater wetlands such as prairie potholes (Van der Valk, 1981). Connell and Slatyer (1977) expanded Gleason's model to include inhibition – early colonizers that "hold their ground" and inhibit colonization by other species, and tolerance – early colonizers tolerate but don't inhibit colonization by other species. In Midwestern wetlands, nutrient enrichment promotes the inhibition model of succession, where aggressive species such as *Typha* and *Phalaris*, colonize and hold the site, inhibiting colonization by other species. *Typha* and *Phalaris* are competitor species according to Grime (1977) and clonal dominants according to Boutin and Keddy (1993). Both species are known to readily colonize disturbed sites such as bare soil (Grace and Harrison, 1986; Green and Galatowitsch, 2002) and once established, they rapidly spread by rhizomes, forming large clonal communities (Wetzel and van der Valk, 1998; Maurer and Zedler, 2002). As *Typha* and *Phalaris* spread, they inhibit other species from colonizing by producing a tall, dense canopy and copious litter that shades the soil and hinders seed germination and plant establishment (Apfelbaum, 1985; Maurer and Zedler, 2002). Nutrient enrichment solidifies the dominance of *Typha* and *Phalaris* as they are able to maximize growth and biomass production relative to other wetland species in response to added nutrients (Maurer and Zedler, 2002; Kercher and Zedler, 2004b).

Litter and soils-based indicators were not strongly related to surface water nutrients. Litter P increased asymptotically with surface water NH_4-N ($r^2 = 0.26$, $p < 0.05$) and total P in soil ($r^2 = 0.26$, $p < 0.01$) was positively related to surface water PO_4-P. In contrast to the Everglades (see review by Noe et al., 2001) and wetlands in Canada (Wisheu et al., 1990) where the vegetation response is linked to P, we did not see strong relationships between soils-based indicators and wetland nutrient condition in our survey of three NEs of the Midwest. The absence of strong relationships is attributed to differences in soil texture among NEs, especially clay, that promotes P sorption and precipitation (Brady and Weil, 2002), and organic matter content. Soils of NE IX contained mostly silt and clay (USDA, 1981a,b) and also had the greatest extractable P and total P concentrations of the three NEs (Table 1). Soil extractable N and total N were greater in NE VII, where soils contained more organic matter, than in NEs VI and IX (Table 1).

Regressions developed for individual NEs generally were more powerful than a single regression for all NEs which was not unexpected since environmental characteristics such as physiography, geology and soils that affect water quality vary among the three NEs. Soils of NE VI (*Corn Belt*), for example, are classified as Mollisols and Alfisols that are relatively young, having formed since the glaciers receded about 10,000 years BP, and weathered from limestone (Buol et al., 1980). These soils are fertile and richer in base cations (Ca, Mg) and P relative to the organic soils (Histosols) of NE VII and the highly weathered Ultisols of NE IX that were not glaciated and, hence, are much older and more weathered and leached (Buol et al., 1980; Cross and Schlesinger, 1995). Thus, high concentration of inorganic P in surface waters of NE VI relative to NEs VII and IX (Table 1) may be partially explained by the fertile soils that underlie this NE.

Differences in anthropogenic land use among NEs though, probably exert a greater influence on wetland nutrient condition than the above mentioned environmental factors. The impact of land use on wetland nutrient and ecological condition are most evident in NE VI (*Corn Belt*) where surface water NH_4-N and PO_4-P concentrations were highest and where few ecological indicators were related to nutrient condition (Table 2). Most land in NE VI is cleared for agriculture (85%) (NCRS, 2004; Tormoehlen et al., 2000) and conversion of native forest to agriculture involves significant disturbance such as drainage of wetlands, tillage practices as well as fertilization. In NEs VII (53% cleared) and IX (24% cleared) where of the landscape is cleared for agriculture (NCRS, 2004; Tormoehlen et al., 2000), vegetation-based indicators were related to surface water NH_4-N (Table 2).

For some indicators, the trajectory or shape of the curve varied from one NE to another. For example, in NEs VII and IX, trajectories of (increasing) aboveground biomass with surface water NH_4-N were distinctly different though both curves were asymptotic (Fig. 4a). Likewise, trajectories of aggressive species biomass (g/m^2) versus surface water N differed for NE VII, where an asymptotic curve best fit the data, versus NEs VI and IX, where linear curves best described the relationship (Fig. 5a).

Vegetation-based indicators, though correlated with surface water N, were not strongly associated with nutrient availability of soils. There was no significant correlation between indices of NPP (aboveground biomass, stem height), species composition (richness, aggressive species) or most measures of nutrient uptake and soil available and total N and P. Only senesced leaf P, which was correlated surface water NH_4-N ($r = 0.57$, $p < 0.01$) and PO_4-P ($r = 0.43$, $p < 0.05$), was correlated with soil extractable P expressed on a mass basis ($r = 0.47$, $p < 0.01$) and a volume basis ($r = 0.44$, $p < 0.05$).

Canonical correlation analysis (CCA) of surface water and plant available (soil extractable) nutrients with species abundance from Appendices A–C supports the regression analysis; that plant community composition is attributed in large part to variation in surface water NH_4-N (Fig. 6). The first CCA axis was positively correlated with surface water NH_4-N ($r = 0.98$) and this axis explained 51% of the variation in the species data. The second axis (29% of the variation) was positively correlated with plant available NH_4-N ($r = 0.55$). Abundance of *Typha* was

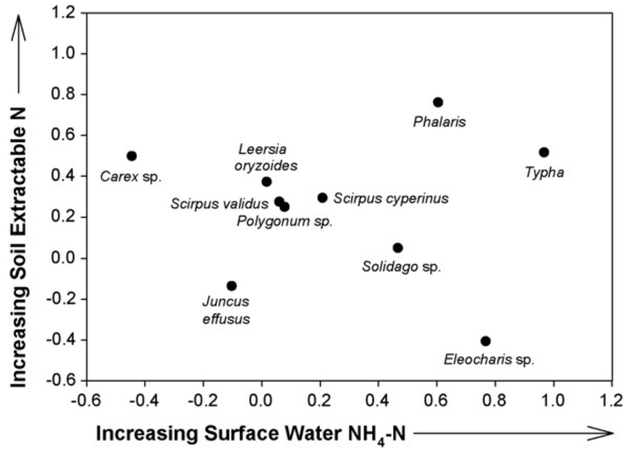


Fig. 6. Canonical correlation analysis of wetland nutrient condition with species abundance for 29 wetlands of NEs VI, VII and IX. Nutrient condition was described by surface water NH_4-N, NO_3-N and PO_4-P and plant available (soil extractable) N and P. Species abundance of the 10 most frequent species (i.e. present at five sites or more) was determined based on their fraction of the total biomass at the site.

associated with high surface water NH_4-N whereas *Phalaris* was associated with high surface water and high plant available N (Fig. 6). Abundance of *Eleocharis* sp. was correlated with high surface water N but low plant available N that was attributed to one site (TNC2) that recently was burned (C.B. Craft, personal observation). And *Carex* sp. was negatively correlated with surface water NH_4-N (Fig. 6). The results of the CCA support experimental and observational studies that link *Typha* and *Phalaris* to nutrient enrichment and *Carex* sp. to low nutrient environments (Wetzel and van der Valk, 1998; Budelsky and Galatowitsch, 2000; Maurer and Zedler, 2002; Woo and Zedler, 2002). It also supports our regression analyses that, for Midwestern wetlands, the response of vegetation is more strongly linked to N concentrations in surface water than in soil.

It is difficult to compare our findings with other NEs around the United States because, in contrast to rivers, streams and lakes, little research of this type has been published for wetlands. The lone exception is the Florida Everglades (NE XIII, Southern Florida Coastal Plain) and, here, the response of wetland vegetation to nutrient enrichment is similar that observed in the Midwest. In the Everglades, vegetation also responds to nutrient enrichment with increased biomass production, stem height and nutrient uptake (Davis, 1991; Miao and Sklar, 1998), decreased species richness and increasing dominance of aggressive species, *T. domingensis* (Jensen et al., 1995; Craft and Richardson, 1997). In contrast to Midwestern wetlands though, the response to nutrient enrichment in the Everglades differs in that (1) the primary limiting nutrient and, thus, the ''problem'' nutrient is P and (2) P enrichment of the underlying peat soils occurs. Everglades soils consist of relatively homogeneous peat that contains abundant nitrogen (2–4%) relative to P (<600 ug/g) (Craft and Richardson, 1993) and, so, strong P limitation of vegetation occurs in this wetland. Furthermore, Everglades soils become phosphorus enriched over time as P enriched detritus from the increasingly nutrient enriched emergent plant community accumulates to produce fresh peat. In contrast, soils of the Midwestern wetlands we sampled vary tremendously within and among NEs, consisting of organic soils (in NE VII only) but more commonly mineral soils, Mollisols, Ultisols, Inceptisols and Entisols, that are low in N and differ in their capacity to retain P. Thus, low soil N content and variable P sorption capacity may explain why, in Midwestern wetlands: (1) N rather than P is linked to

Table 3
Proposed ecological indicators of wetland nutrient condition for Nutrient Ecoregions (NEs) of the Midwest

Indicator	Model	NE	Goodness of fit (r^2), p value
Aboveground biomass (g/m^2)	$1096 \times (1 - e^{-23(NH_4\text{-}N)})$	VII + IX	$r^2 = 0.36$, $p < 0.01$
Stem height (cm)	$167 \times (1 - e^{-28(NH_4\text{-}N)})$	VII + IX	$r^2 = 0.56$, $p < 0.0002$
Species richness (no./site)	$12.33–33 \times (NH_4\text{-}N)$	VII + IX	$r^2 = 0.33$, $p < 0.01$
Aggressive species (g/m^2)	$620 \times (1 - e^{-11(NH_4\text{-}N)})$	VII	$r^2 = 0.51$, $p < 0.05$
	$-128 + 8920 \times (NH_4\text{-}N)$	VI + IX	$r^2 = 0.58$, $p < 0.0005$
Aggressive species (%)	$106 \times (1 - e^{-7(NH_4\text{-}N)})$	VII + IX	$r^2 = 0.54$, $p < 0.0002$

Regression models were chosen based on the significance (p value) level.

nutrient enrichment and (2) P enrichment of the soil is not evident.

For Midwestern wetlands, the best indicators and their regression models, based on the significance level (p value), are shown in Table 3. Overall, stem height and percent biomass of aggressive species were the best indicators of nutrient condition. They had high goodness of fit (0.54–0.56), low p values (<0.0002) and were applicable to NEs VII and IX. Also, in NE VII and IX, there appeared to be a threshold concentration, above which *Typha* and *Phalaris* dominate. In wetlands where surface water NH_4-N exceeded 40 ug/L, these species accounted for more than 40% of total plant biomass (see Fig. 5b). Species richness ands aboveground biomass also were robust indicators that were applicable to NEs VII and IX but the goodness of fit was not very high (0.33–0.36). Biomass (g/m^2) of aggressive species was not robust because the best models were derived for individual, not multiple NEs.

We conclude that indices of vegetation NPP and species composition are robust indicators of nutrient condition of freshwater wetlands of the Midwestern U.S., especially in NEs where other anthropogenic disturbances (e.g. land clearing, drainage) are not widespread and intense. Soils-based indicators are less effective than vegetation because properties such as texture and organic matter that affect soil nutrient enrichment vary so much among NEs. Additional work is needed to test these indicators across a range of NEs, wetland vegetation types and human disturbance regimes.

Acknowledgements

We appreciate the help of many students (Scott Struck, Kristie Overberg, Chad Washburn, Christina Pruett, Sarah Butler, Angela Vedder) who participated in field sampling and lab analyses and the land owners (Foxwood Farms, The Nature Conservancy-Kankakee Sands Preserve, Indiana Department of Natural Resources) who gave us permission to collect samples on their property. We gratefully acknowledge funding support from U.S. EPA Region 5 (Chicago) and U.S. EPA Headquarters. We appreciate the thoughtful comments of Paul McCormick, Ulo Mander and an anonymous reviewer who examined an earlier draft the manuscript.

Appendix A

Plant species collected from 10 freshwater wetlands located in Nutrient Ecoregion VI. Values in parentheses are percent of total aboveground biomass for a given site.

LA1	LA2	MOR	TNC1	TNC2	TNC3	TNC4	WS1	WS2	WS3
Polygonum sp. (63)	*Lemna* spp. (<1)	*Typha latifolia* (100)	*Eleocharis* sp. (4)	*Bidens cernua* (<1)	*T. latifolia* (100)	*Amaranthus* spp. (<1)	*T. latifolia* (100)	*Acer rubra* (<1)	*Bidens connata* (<1)
Sagittaria latifolia (1)	*Nymphaea odorata* (4)	Unknown grass (<1)	*Leersia oryzoides* (2)	*Bidens coronata* (<1)		*Brassicaceae* (<1)	Unknown forb (<1)	*Boehmeria cylindrical* (1)	*Echinochlea muricata* (<1)
Scirpus validus (8)	*T. latifolia* (96)		*Mimulus ringens* (5)	*Eleocharis* sp. (59)		*Equisetum laevigatum* (20)		*T. latifolia* (87)	*Eleocharis* sp. (<1)
T. latifolia (28)			*Polygonum* sp. (2)	*L. oryzoides* (10)		*Lycopus* sp. (1)		Unknown grass #1 (<1)	*L. oryzoides* (1)
Utricularia spp. (<1)			*Salix* sp. (5)	*M. ringens* (4)		*Panicum* sp. (2)		Unknown grass #2 (12)	*Polygonum* sp. (19)
			S. pungens (27)	*Schoenoplectus pungens* (3)		*S. pungens* (35)		Unknown forb (<1)	*Phalaris arundinacea* (6)
			S. validus (54)	*Scirpus fluviatilis* (5)		*Salix* spp. (1)			*S. latifolia* (1)
			T. latifolia (1)	*S. validus* (7)		*T. latifolia* (41)			*T. latifolia* (73)
			Unknown forb (<1)	*T. latifolia* (12)					Unknown grass (<1)
									Unknown forb (<1)

Appendix B

Plant species collected from 10 freshwater wetlands located in Nutrient Ecoregion VII. Values in parentheses are percent of total aboveground biomass for a given site.

FM	MIT	MIT2	NF	NFS	NOT	PR	RM	TF	WOL
Bidens sp. (19)	*B. cernua* (14)	*Carex* sp. (2)	*A. rubra* (<1)	*B. cylindrica* (1)	*B. cernua* (14)	*Apios americana* (1)	*Carex* sp. (6)	*Asclepias incarnata* (<1)	*B. cernua* (<1)
Carex sp. (26)	*Cyperus esculentus* (19)	*Echinochloa crusgalli* (<1)	*Betula pumila* (4)	*C. mariscoides* (1)	*Eleocharis* sp. (3)	*Asclepias incarnata* (3)	*Eupatorium maculatum* (1)	*Cladium mariscoides* (57)	*Eleocharis* sp. (2)
L. oryzoides (<1)	*Echinochloa waltri* (20)	*Eleocharis* sp.	*C. mariscoides* (38)	*Carex* sp. (21)	*Polygonum puncatum* (<1)	*Carex* sp. (41)	*Onoclea sensibilis* (20)	*Carex* sp. (1)	*Juncus effusus* (<1)
P. arundinacea (35)	*L. oryzoides* (33)	*L. oryzoides*	*Carex* sp. (23)	*Eleocharis* sp. (2)	*S. latifolia* (2)	*E. maculatum* (9)	*P. arundinacea* (44)	*Eleocharis* sp. (3)	*L. oryzoides* (10)
Solidago sp. (6)	*P. hydropiperoides* (10)	*S. latifolia* (1)	*Drosera reutundifolia* (<1)	*Juncus effusus* (1)	*T. latifolia* (81)	*O. sensibilis* (6)	*Polygonum sagittatum* (<1)	*Equisetum* sp. (<1)	*Polygonum* sp. (<1)
Solidago uliginosa (3)	*T. latifolia* (4)	*Scirpus americanus* (22)	*Eleocharis* sp. (23)	*Juncus* sp. (<1)		*Pedicularis lanceolata* (1)	*Polygonum* sp. (<1)	*Hypericum* sp. (<1)	*P. arundinacea* (5)
T. latifolia (11)		*S. validus* (1)	*E. laevigatum*	*Lathyrus palustris* (<1)		*Phalaris arundicacea* (15)	*Rosa palustris* (<1)	*J. effusus* (1)	*T. latifolia* (83)
		T. latifolia (74)	*Juncus brachycephalus* (2)	*L. oryzoides* (<1)		*Polygonum convolvulus* (<1)	*S. americanus* (26)	*Juncus* sp. (<1)	
			Juncus sp. (<1)	*Lycopus* sp. (<1)		*P. sagittatum* (<1)	*Solidago* sp. (2)	*Potentilla fruticosa* (36)	
			L. oryzoides (<1)	*Mentha arvensis* (2)		*S. americanus* (4)	*Thelypteris palustris* (<1)	*P. lanceolata* (1)	
			Lobelia cardinalis (<1)	*P. arundinacea* (5)		*Solidago gigantea* (3)		*Setaria viridis* (<1)	
			Pedicularis lanceolate	*Rosa* sp. (<1)		*Solidago* sp. (14)		*Scirpus* sp. (<1)	
			P. fruticosa (2)	*S. americanus* (19)		*Thalictrum revolutum* (1)		*Solidago* sp. (1)	
			Rudbeckia sp. (2)	*Scirpus* sp. (1)		*T. palustris* (1)			
			Solidago sp. (<1)	*S. validus* (8)					
			S. uliginosa (<1)	*Salix nigra* (13)					
			S. validus (4)	*T. palustris* (4)					
				T. latifolia (22)					

Appendix C

Plant species collected from 10 freshwater wetlands located in Nutrient Ecoregion IX. Values in parentheses are percent of total aboveground biomass for a given site.

AR	BOT	BV	GLL	GLU	LM1	LM2	LS	SYC	TL
J. effusus (32)	*L. oryzoides* (4)	*Carex* sp. (<1)	*B. cernua* (5)	*Acer saccharinum* (2)	*B. cylindrica* (1)	*Bidens coronata* (1)	*B. cernua* (<1)	*Erigeron* spp. (3)	*B. cylindrica* (<1)
Juncus tenuis (<1)	*T. latifolia* (91)	*Juncus* sp. (<1)	*J. effusus* (3)	*B. cylindrica* (2)	*L. oryzoides* (2)	*B. connata* (11)	*B. connata* (4)	*J. effusus* (3)	*Impatiens capensis* (3)
L. oryzoides (22)	Unknown grass (5)	*J. tenuis* (1)	*L. oryzoides* (2)	*J. effusus* (3)	*P. sagittatum* (4)	*C. esculentus* (4)	*J. effusus* (17)	*L. oryzoides* (21)	*Lysimachia nummularia* (<1)
L. nummularia (<1)		*L. oryzoides* (19)	*Lamium* sp. (1)	*L. oryzoides* (1)	*Polygonum* sp. (26)	*Eleocharis* sp. (4)	*L. oryzoides* (20)	*Scirpus cyperinus* (39)	*S. cyperinus* (3)
Scirpus atrivirens (20)		*S. cyperinus* (<1)	*Scirpus atrovirens* (39)	*L. nummularia* (8)	*S. cyperinus* (32)	*Eragrostis hypnoides* (2)	*Polygonum* sp. (2)	*T. latifolia* (13)	*T. latifolia* (94)
Scirpus cyperinuns (12)		*T. latifolia* (80)	*S. cyperinus* (<1)	*P. sagittatum* (5)	Unknown forb #1 (<1)	*L. oryzoides* (37)	*P. arundinacea* (2)	Unknown forb #1 (17)	Unknown grass (<1)
T. latifolia (3)		Unknown grass (<1)	*T. latifolia* (50)	*S. atrovirens* (<1)	Unknown forb #2 (11)	*Polygonum* sp. (19)	*T. latifolia* (55)	Unknown forb #2 (1)	Unknown forb (<1)
Unknown grass (4)		Un. forb #1 (<1)	Unknown grass (<1)	*S. cyperinus* (11)	Unknown forb #3 (9)	*S. cyperinus* (17)	Unknown grass (<1)	Unknown forb #3 (1)	
Unknown forb #1 (2)		Unknown forb #2 (<1)		*T. latifolia* (67)	*Xanthium strumarium* (15)	Unknown forb (<1)	Unknown forb #1 (<1)	Unknown forb #4 (2)	
Unknown forb #2 (3)		Unknown forb #3 (<1)		Unknown forb #1 (<1)		*X. strumarium* (5)	Unkown forb #2 (<1)	Unknown forb #5 (<1)	
Unknown forb #3 (1)		Unknown forb #4 (<1)		Unknown forb #2 (<1)			Unknown forb #3 (<1)		
Unknown forb #4 (1)				Unknown forb #3 (<1)					
Unknown forb #5 (<1)									

References

Aerts, R., Berendse, F., 1988. The effect of increased nutrient availability on vegetation dynamics in wet heathlands. Vegetation 76, 63–69.

Amon, J.P., Thompson, C.A., Carpenter, Q.J., Miner, J., 2002. Temperate zone fens of the glaciated Midwestern USA. Wetlands 22, 301–317.

Apfelbaum, S., 1985. Cattail (*Typha* spp.) management. Nat. Areas J. 5, 9–17.

APHA (American Public Health Association), 1998. Standard Methods for the Examination of Water and Wastewater. American Public Health Association, Washington, DC.

Boutin, C., Keddy, P.A., 1993. A functional classification of wetland plants. J. Veg. Sci. 4, 591–600.

Brady, N.C., Weil, R.R., 2002. The Nature and Properties of Soils. Prentice Hall, Upper Saddle River, NJ, p. 960.

Bridgham, S.D., Pastor, J., Jannsens, J.A., Chapin, C., Malterer, T.J., 1996. Multiple limiting gradients in peatlands: a call for a new paradigm. Wetlands 16, 45–65.

Budelsky, R.A., Galatowitsch, S.M., 2000. Effects of water regime and competition on the establishment of a native sedge in restored wetlands. J. Appl. Ecol. 37, 971–985.

Buol, S.W., Hole, F.D., McCracken, R.J., 1980. Soil Genesis and Classification. Iowa State University Press, Ames, Iowa, p. 406.

Chambers, R.M., Meyerson, L.A., Saltonstall, K., 1999. Expansion of *Phragmites australis* into tidal wetlands of North America. Aquat. Bot. 64, 261–273.

Childers, D.L., Doren, R.F., Jones, R., Noe, G.B., Rugge, M., Scinto, L.J., 2003. Decadal change in vegetation and soil phosphorus patterns across the Everglades landscape. J. Environ. Qual. 32, 344–362.

Clements, F.E., 1928. Plant Succession and Indicators. H.W. Wilson, New York, NY.

Connell, J.H., Slatyer, R.O., 1977. Mechanisms of succession in natural communities and their role in community stability and organization. Am. Nat. 111, 1119–1144.

Craft, C.B., Seneca, E.D., Broome, S.W., 1991. Porewater chemistry of natural and created marsh soils. J. Exp. Mar. Biol. Ecol. 152, 187–200.

Craft, C.B., Richardson, C.J., 1993. Peat accretion and phosphorus accumulation along an eutrophication gradient in the northern Everglades. Biogeochemistry 22, 133–156.

Craft, C.B., Richardson, C.J., 1997. Relationships between soil nutrients and plant species composition in Everglades's peatlands. J. Environ. Qual. 26, 224–232.

Cross, A.M., Schlesinger, W.J., 1995. A literature review and evaluation of the Hedley fractionation: applications to the biogeochemical cycle of soil phosphorus in natural ecosystems. Geoderma 64, 197–214.

Davis, S.M., 1991. Growth, decomposition and nutrient retention of *Cladium jamaicense* Crantz and *Typha domingensis* Pers. in the Florida Everglades. Aquat. Bot. 40, 203–224.

DeBusk, W.F., Reddy, K.R., Koch, M.S., Wang, Y., 1994. Spatial distribution of soil nutrients in a northern Everglades marsh: water conservation area 2A. Soil Sci. Soc. Am. J. 58, 543–558.

DeBusk, W.F., Newman, S., Reddy, K.R., 2001. Spatio-temporal patterns of soil phosphorus enrichment in Everglades water conservation area 2A. J. Environ. Qual. 30, 1438–1446.

Doren, R.F., Armentano, T.V., Whiteaker, L.D., Jones, R.D., 1999. Marsh vegetation patterns and soil phosphorus gradients in the Everglades ecosystem. Aquat. Bot. 56, 145–163.

Drexler, J.Z., Bedford, B.L., 2002. Pathways of nutrient loading and impacts on plant diversity in a New York peatland. Wetlands 22, 263–281.

Fassett, N.C., 1957. A manual of Aquatic Plants. University of Wisconsin Press, Madison, WI, p. 416.

Galatowitsch, S.M., Anderson, N.O., Ascher, P.D., 1999. Invasiveness in wetland plants in temperate North America. Wetlands 19, 733–755.

Gerloff, G.C., Krombholz, P.H., 1966. Tissue analysis as a measure of nutrient availability for the growth of angiosperm aquatic plants. Limnol. Oceanogr. 11, 529–539.

Gleason, H.A., 1927. Further views on the succession concept. Ecology 8, 299–326.

Gleason, H.A., Cronquist, A., 1991. Manual of Vascular Plants of Northeastern United States and Adjacent Canada. D. Van Norstrand Company, New York, p. 986.

Grace, J.B., Harrison, J.S., 1986. The biology of Canadian weeds 73. *Typha latifolia* L., *Typha angustifolia* L. and *T xglauca* Godr. Can. J. Plant Sci. 66, 361–369.

Green, E.K., Galatowitsch, S.M., 2002. Effects of *Phalaris arundinacea* and nitrate-N addition on the establishment of wetland plant communities. J. Appl. Ecol. 39, 134–144.

Grime, J.P., 1977. Evidence for the existence of three primary strategies in plants and its relevance to ecological and evolutionary theory. Am. Nat. 111, 1169–1194.

Gusewell, S., Koerselman, W., Verhoeven, J.T.A., 2003. Biomass N:P ratios as indicators of nutrient limitation for plant populations in wetlands. Ecol. Appl. 13, 372–384.

Gustafson, S., Wang, D., 2002. Effects of agricultural runoff on vegetation composition of a priority conservation wetland, Vermont, USA. J. Environ. Qual. 31, 350–357.

Hayati, A.A., Proctor, M.C.F., 1991. Limiting nutrients in acid-mire vegetation: peat and plant analyses and experiments on plant responses to added nutrients. J. Ecol. 79, 75–95.

Jensen, J.R., Rutchey, K., Koch, M.S., Narumalani, S., 1995. Inland wetland change detection in the Everglades water conservation area 2A using a time series of normalized remotely sensed data. Photogramm. Eng. Rem. Sens. 61, 199–209.

Kercher, S.M., Zedler, J.B., 2004a. Flood tolerance in wetland angiosperms: a comparison of invasive and noninvasive species. Aquat. Bot. 80, 89–102.

Kercher, S.M., Zedler, J.B., 2004b. Multiple disturbances accelerate invasion of reed canary grass (*Phalaris arundinacea* L.) in a mesocosm study. Oecologia 138, 455–464.

Koch, M.S., Reddy, K.R., 1992. Distribution of soil and plant nutrients along a trophic gradient in the Florida Everglades. Soil Sci. Soc. Am. J. 56, 1492–1499.

Koerselman, W., Mueleman, A.F.M., 1996. The vegetation N:P ratio: a new tool to detect the nature of nutrient limitation. J. Appl. Ecol. 33, 1441–1450.

Kuo, S., 1996. Phosphorus. In: Sparks, D.L., et al. (Eds.), Methods of Soil Analysis Part 3: Chemical Methods. SSSA Book Series 5. American Society of Agronomy, Madison, WI, pp. 869–920.

Maurer, D.A., Zedler, J.B., 2002. Differential invasion of a wetland grass explained by tests of nutrients and light availability on establishment and clonal growth. Oecologia 131, 279–288.

McCormick, P.V., Rawlik, P.S., Lurding, K., Smith, E.P., Sklar, F.H., 1996. Periphyton–water quality relationships along a nutrient gradient in the northern Florida Everglades. J. North Am. Benthol. Soc. 15, 433–449.

Miao, S.L., Sklar, F.H., 1998. Biomass and nutrient allocation of sawgrass and cattail along a nutrient gradient in the Florida Everglades. Wetlands Ecol. Manag. 5, 245–263.

Mulvaney, R.L., 1996. Nitrogen–inorganic forms. In: Sparks, D.L., et al. (Eds.), Methods of Soil Analysis, Part 3. SSSA Books Ser 5. SSSA, Madison, WI, pp. 1123–1184.

Newman, S.M., Grace, J.B., Koebel, J.W., 1996. Effects of nutrients and hydroperiod on *Typha*, *Cladium* and *Eleocharis*: implications for Everglades's restoration. Ecol. Appl. 6, 774–783.

Noe, G.B., Childers, D.L., Jones, R.D., 2001. Phosphorus biogeochemistry and the impact of phosphorus enrichment: why is the Everglades so unique? Ecosystems 4, 603–624.

North Central Research Station, 2004. The Changing Midwest Assessment. USDA Forest Service, Rhinelander, WI.

Odum, E.P., Finn, J.T., Franz, E.H., 1979. Perturbation theory and the subsidy-stress gradient. BioScience 29, 349–352.

Omernick, J.M., 1987. Ecoregions of the conterminous United States. Map (scale 1:7,500,00). Ann. Assoc. Am. Geographers 77, 118–125.

Pegtel, D.M., Bakker, J.C., Verweij, G.L., Fresco, L.F.M., 1996. N, K and P deficiency in chronoseqential cut summer-dry grasslands on gley podzol after cessation of fertilizer application. Plant Soil 178, 121–131.

Ponnamperuma, F.C., 1972. The chemistry of submerged soils. Adv. Agron. 24, 29–96.

Rawlings, J.O., Pantula, S.G., Dickey, D.A., 2001. Applied Regression Analysis: A Research Tool. Springer-Verlag, New York, 675 pp.

Qualls, R.G., Richardson, C.J., 1995. Forms of soil phosphorus along a nutrient enrichment gradient in the northern Everglades. Soil Sci. 160, 183–198.

Qualls, R.G., Richardson, C.J., 2000. Phosphorus enrichment affects litter decomposition, immobilization and soil microbial phosphorus in wetland mesocosms. Soil Sci. Soc. Am. J. 64, 799–808.

SAS (Statistical Analysis Systems), 2002. The SAS 9.1 system for Windows. SAS Institute Inc., Cary, NC.

Shaver, G.R., Chapin III, F.S., 1995. Long-term responses to factorial NPK fertilizer treatment by Alaskan wet and moist tundra sedge species. Ecography 18, 259–275.

Shaver, G.R., Johnson, L.C., Cades, D.H., Murray, G., Laundre, J.A., Rastetter, E.B., Nadelhoffer, K.J., Giblin, A.E., 1998. Biomass and CO_2 flux in wet sedge tundras: responses to nutrients, temperature and light. Ecol. Monogr. 68, 75–97.

Shaver, G.R., Bret-Harte, M.S., Jones, M.H., Johnstone, J., Gough, L., Laundre, J., Chapin III, F.S., 2001. Species composition interacts with fertilizer to control long-term change in tundra productivity. Ecology 82, 3163–3181.

Sommers, L.E., Nelson, D.W., 1972. Determination of total phosphorus in soils: a rapid perchloric acid digestion procedure. Soil Sci. Soc. Am. Proc. 36, 902–904.

Svengsouk, L.J., Mitsch, W.J., 2001. Dynamics of mixtures of *Typha latifolia* and *Schoenoplectus tabernaemontani* in nutrient enrichment wetland experiments. Am. Midland Nat. 145, 309–324.

SYSTAT Software, 2004. SigmaPlot [Computer Software]. Systat Software, Inc., Point Richmond, CA.

Tormoehlen, B., Gallion, J., Schmitt, T.L., 2000. Forests of Indiana: 1998 Overview. NA-TP-03-00. USDA (U.S. Department of Agriculture) Forest Service, 18 pp.

USDA (U.S. Department of Agriculture), 1981a. Soil survey of Monroe County, Indiana. U.S. Soil Conservation Service, Washington, DC.

USDA (U.S. Department of Agriculture), 1981b. Soil survey of Steuben County, Indiana. U.S. Soil Conservation Service, Washington, DC.

USDA (U.S. Department of Agriculture), 1998. Soil survey of Newton County, Indiana. U.S. Soil Conservation Service, Washington, DC.

U.S. EPA, 2002. Methods for Evaluating Wetland Nutrient Condition: #16 Vegetation-based Indicators of Wetland Nutrient Enrichment. EPA-822-R-02-024. Office of Water, U.S. Environmental Protection Agency, Washington, DC.

Van der Valk, A.G., 1981. Succession in wetlands: a Gleasoniam approach. Ecology 62, 688–696.

Venterink, H.O., Pieterse, N.M., Belgers, D.M., Wassen, M.J., De Ruiter, P.C., 2002. N, P and K budgets along nutrient availability and productivity gradients in wetlands. Ecol. Appl. 12, 1010–1024.

Verhoeven, J.T.A., Schmitz, M.B., 1991. Control of plant growth by nitrogen and phosphorus in mesotrophic fens. Biogeochemistry 12, 135–148.

Verhoeven, J.T.A., Koerselman, W., Meuleman, A.F.M., 1996. Nitrogen or phosphorus limited growth in herbaceous wet vegetation: relations with atmospheric inputs and management regimes. Trends Ecol. Evol. 11, 494–497.

Vermeer, H.J.G., 1986. The effect of nutrients on shoot biomass and species composition of wetland and hayfield communities. Acta Oecologica/Oecologia Plantarum 7, 31–41.

Voss, E.G., 1996a. Michigan Flora—Part I Gymnosperms and Monocots. Bulletin Number 55. Cranbrook Institute of Science, Bloomfield Hills, Michigan and University of Michigan Herbarium, Ann Arbor, MI.

Voss, E.G., 1996b. Michigan Flora—Part III Dicots (Pyrolaceae–Compositae). Bulletin Number 61. Cranbrook Institute of Science, Bloomfield Hills, Michigan and University of Michigan Herbarium, Ann Arbor, MI.

Wetzel, P.R., van der Valk, A.G., 1998. Effects of nutrients and soil moisture on competition between *Carex stricta*, *Phalaris arundinacea* and *Typha latifolia*. Plant Ecol. 138, 179–190.

Willis, A.J., 1963. Braunton burrows: the effects on the vegetation of the addition of mineral nutrients to the dune soils. J. Ecol. 51, 353–374.

Wisheu, I.C., Keddy, P.A., Moore, R.D.R.J., McCanny, S.J., Gaudet, C.L., 1990. Effects of eutrophication on wetland vegetation. In: Kusler, J., Smardon, R. (Eds.), Wetlands of the Great Lakes: Protection and Restoration Policies; Status of the Science. Symposium Proceedings, Niagara, New York, May. Association of State Wetlands Managers Inc., pp. 112–121.

Woo, I., Zedler, J.B., 2002. Can nutrients alone shift a sedge meadow towards dominance by the invasive *Typha* × *glauca*? Wetlands 22, 509–521.

Wright, A.L., Reddy, K.R., 2001. Heterotrophic microbial activity in northern Everglades's wetland soils. Soil Sci. Soc. Am. J. 65, 1856–1864.

WETLANDS, Vol. 27, No. 2, June 2007, pp. 240–250

ECOSYSTEM GAS EXCHANGE ACROSS A CREATED SALT MARSH CHRONOSEQUENCE

Jeff A. Cornell[1], Christopher B. Craft[2], and J. Patrick Megonigal[3]
[1]*George Mason University*
4400 University Drive
Fairfax, Virginia, USA 22030

[2]*School of Public and Environmental Affairs*
Indiana University
Bloomington, Indiana, USA 47405

[3]*Corresponding author: Smithsonian Environmental Research Center*
P.O. Box 28, 647 Contees Wharf Road
Edgewater, Maryland, USA 21037-0028
E-mail: megonigalp@si.edu

Abstract: Salt marshes created on dredge spoil were compared to natural marshes to evaluate the capacity of created marshes to perform carbon cycle functions. Several carbon cycle attributes were measured in eight created *Spartina alterniflora* Loisel salt marshes that ranged from one to 28 years, each paired with a nearby mature natural reference marsh. The attributes measured included gross primary production, respiration, net ecosystem exchange, potential microbial respiration (CH_4 and CO_2), and aboveground biomass. *In situ* exchange rates of CO_2 and plant biomass in created marshes met or exceeded those of reference marshes in three to four years. There was some evidence that ecosystem gas exchange in created marshes developed slightly faster than aboveground biomass production. Soil carbon mineralization per gram carbon was generally higher in the created marshes than reference marshes, suggesting higher carbon quality and/or nutrient availability in the created marshes. However, carbon mineralization rates per gram soil were relatively low in the created marshes due to lower soil organic matter content. With proper construction, we suggest most major carbon fluxes can be established in created salt marshes in less than five years.

Key Words: carbon cycle, created marsh, gas exchange, *Spartina alterniflora*, succession

INTRODUCTION

Wetland creation in the U.S. is a common strategy to mitigate wetland losses due to draining and filling (LaSalle et al. 1991, Noon 1996, Shafer and Streever 2000) or to stabilize sediments (Seneca et al. 1985, Broome et al. 1988). Numerous salt marshes have been created on sandy, low organic matter substrates dredged from North Carolina's Intercoastal Waterway since about 1971 (Craft et al. 1999). These projects involved the creation of large dredge spoil-islands that were later graded and planted with appropriate marsh plant species, such as *Spartina alterniflora* Loisel and *S. patens* (Aiton) Muhl (Radford et al. 1968), which stabilize the dredge material. It is often assumed that a young salt marsh created on dredge spoil substrate is functionally equivalent to a natural salt marsh of a comparable age, and that created marshes have the potential to replace the functions of mature natural salt marshes given enough time (Langis et al. 1991, Craft 1997, Craft et al. 1999). However, few studies have been conducted to test these assumptions (Poach and Faulkner 1998).

Carbon is arguably the most fundamental element to quantify when assessing the pace of ecosystem development. Most wetland services are influenced directly or indirectly by the capacity of the ecosystem to produce, process, and store organic carbon (Craft et al. 1988a, Craft et al. 2003). Gross primary production (GPP) largely establishes the upper limit of heterotrophic activity in the system, including the secondary productivity of consumers. Microbial respiration (R) and decomposition of organic matter either releases or sequesters nitrogen (and other nutrients), depending on the chemical characteristics of the detritus. Labile carbon availability influences rates of microbial redox transformations such as denitrification. The capacity of an ecosystem to sequester atmospheric CO_2 in biomass or soil organic matter represents an imbalance

Table 1. Characteristics of created marshes, including their age when the study began and ended.

Marsh Name	Site	Age (yr)	Substrate	County (NC)	Year Planted	Salinity (ppt)	Tidal Range (m)
DOT	Y1	1–2	Dredge Spoils	Carteret	1997	20–30	1.0
Consultant	Y3	3–4	Dredge Spoils	Carteret	1996	17–32	1.0
Port	Y8	8–9	Dredge Spoils	Carteret	1990	18–30	1.0
Swansboro	Y11	11–12	Dredge Spoils	Onslow	1987	20–30	1.1
Dill's Creek	Y13	13–14	Graded-Upland	Carteret	1985	14–33	1.0
Pine Knoll	Y24	24–25	Dredge Spoils	Carteret	1974	20–30	1.0
Marine Lab	Y26	26–27	Dredge Spoils	Carteret	1972	20–30	1.0
Snow's Cut	Y28	28–29	Dredge Spoils	New Hanover	1970	5–20	1.2

between GPP and R (i.e., net ecosystem production, NEP), and the net exchange of particulate and dissolved organic carbon with adjacent ecosystems.

Most previous studies of created wetlands have focused on changes in the size of key carbon pools such as soil organic matter (Craft et al. 1988a, b, Craft et al. 1989, Dayton et al. 1996, Padgett and Brown 1999, Streever 2000), organically bound soil nutrients (Langis et al. 1991, Craft 1997), and plant biomass (Broome et al. 1986, Craft et al. 1999). Based largely on changes in pool sizes and pool accumulation rates, Craft et al. (2003) proposed a conceptual model consisting of three distinct trajectories of ecosystem development in a chronosequence of created *S. alterniflora* marshes ranging from one to 28 years old. The ecological attributes that developed most rapidly, such as sediment and particulate carbon accumulation, were linked directly to the successful establishment of hydrology. Biological attributes such as plant biomass required five to 15 years to converge on the range expected for natural salt marshes, and more than 28 years was required for pools of soil organic matter to reach natural marsh levels. Similar patterns have been observed in other studies of soil carbon pools (Craft 1988b, Langis et al. 1991, Craft 1997, Craft 1999).

Although some previous studies of created wetlands evaluated aspects of carbon cycling, none focused on the full suite of processes that constitute the ecosystem carbon cycle. In particular, the key processes of GPP, R, and net ecosystem exchange (NEE) of carbon dioxide have been overlooked. Immature marshes with low biomass can be expected to have relatively low GPP and R compared to mature marshes with high biomass. As created marshes age, GPP, R, and NEE should increase to levels that meet or exceed those characteristic of mature ecosystems (Odum 1969). We evaluated development of these relatively dynamic features of the carbon cycle in a chronosequence of created *S. alterniflora* marshes. Our goal was to determine the amount of time required for carbon pools and gas exchange rates in created salt marshes to reach parity with natural marshes. In particular, we tested the hypothesis that GPP, R, and NEE would be lower in newly created marshes than in nearby natural marshes, but the performance of created marshes would increase to meet or exceed the natural marshes.

MATERIALS AND METHODS

Study Sites

We studied eight created marshes in coastal North Carolina that ranged in age from two to 29 years as of November 1999 (Table 1). Each created marsh was paired with a nearby natural marsh to allow comparisons between created and natural marshes of similar hydrology, salinity, temperature, and other location-specific parameters (e.g., Conn and Day 1997, Craft 1997). Natural marshes were assumed to represent created marshes in a late stage of development. Each marsh experiences high tides twice daily with tidal ranges of approximately 1 m.

The created marshes were generally established on sandy dredge spoil islands, with the sole exception of Dill's Creek marsh, which was created on graded, upland soil. All of the created marshes were originally planted with *S. alterniflora*, which still dominated the plant community at the time of the study. Sampling took place within the *S. alterniflora* community along 30 m-long transects oriented parallel to the closest shoreline or tidal creek, approximately 10 m inland where practical.

Net Ecosystem Exchange

NEE of CO_2 was measured using static chambers. The frames for two static chambers were constructed from 2.5-cm-wide angle aluminum. The base dimensions were 0.5 m $\times$ 0.5 m, and the heights were either 0.9 m or 1.5 m to accommodate different vegetation heights. Closed-cell foam was attached to the bases to insure an airtight seal between the chamber bottom and collars, which were permanently installed in the soil to a depth of 5 cm. Chamber walls were constructed of Tefzel (DuPont, Inc., Circleville, Ohio), which is a clear sheeting that is impermeable to gases. Air temperature changes in the chamber were less than 1°C during incubations.

One month prior to our first sampling campaign, five aluminum collars were randomly placed along a 30-m transect at each marsh. The minimum distance between collars was 3 m. Chambers were clamped to the collars to maintain an airtight seal.

A LI-COR 6200 Portable Infrared Gas Analyzer (herein IRGA) was connected with Bev-a-Line tubing (Thermoplastic Processes, Inc., Stirling, New Jersey) to the chambers for CO_2 measurement. The IRGA was run in closed mode so that chamber air was replaced after sampling. Tubing inside the chamber was oriented to avoid re-sampling. Brushless electric fans were positioned near the chamber base and top to circulate chamber air.

CO_2 fluxes were measured quarterly five times over 13 months beginning in July 1998. Respiration was measured in July and October 1998, as well as January, March, and July 1999. NEE and GPP were measured in July 1998 and July 1999. Incubation time was generally less than 180 s. NEE was measured in full sunlight, which was generally greater than 1,000 μmol m^{-2} s^{-1} and stable during the incubation. Using the same incubation procedure as described above, respiration rates were determined by placing an opaque cover over the chamber. GPP rates were calculated as:

$$\mathrm{GPP} = \mathrm{NEE} + \mathrm{R} \quad (1)$$

Note that both GPP and R are reported as positive values with the understanding that the actual carbon fluxes are in opposite directions relative to the atmosphere.

In July 1999, light response curves were generated by measuring NEE at five different light levels. Light levels were manipulated by placing layers of nylon window screening over the chamber. Two light curves were generated at each marsh.

To obtain a more robust estimate of NEE over a typical summer day (July 1999), we modeled daily NEE using empirical relationships between photosynthetically active radiation (PAR) and temperature (Morris and Whiting 1986, Neubauer et al. 2000). The model was driven by PAR and temperature data measured at the Institute of Marine Sciences in Morehead City, North Carolina (IMS 1999), which was within 140 km of all sites and within 10 km of most sites.

GPP was modeled as a hyperbolic function of light on an hourly time step:

$$\mathrm{GPP} = [(a * I)/(b + I)] \quad (2)$$

where I is average hourly photon flux and a and b are empirically derived constants with units of μmol C m^{-2} h^{-1} and μE m^{-2} h^{-1}, respectively (Neubauer 2000). Hourly temperature data was used to calculate R rates (n = 2) as:

$$R_t = R_0 \times Q_{10}{}^\wedge[(T_t - T_0)/10] \quad (3)$$

where R_t is the calculated respiration rate (μmol C m^{-2} h^{-1}), R_0 is the initial ecosystem respiration rate, Q_{10} is the temperature coefficient, T_0 is the initial air temperature (°C), and T_t is the air temperature at one of the hourly time steps. Hourly GPP and R rates were summed to obtain daily rates (Neubauer 2000). Q_{10} was calculated as:

$$\begin{aligned} \log Q_{10} &= \log(k_2/k_1) \\ &= (E_a/2.3R) \times 10/(T_2 \times T_1) \end{aligned} \quad (4)$$

where E_a is the activation energy, R is the gas constant, k_1 and k_2 are reaction rates, T_1 and T_2 are air temperatures (ideally a 10°K temperature difference, °K), and the term $E_a/2.3R$ was determined from the slope of an Arrhenius plot of log(k) versus 1/temperature (Segel 1976).

Live and standing-dead stem and leaf material was removed from inside the aluminum collars in November 1999. Plant material was returned to the laboratory and dried at 70°C to constant weight. Biomass data from these marshes was previously reported by Craft et al. (2003). However, the Craft et al. (2003) data were collected in October 1998 from locations several meters away from our gas flux plots. The biomass data reported in the present study were collected from the field CO_2 exchange plots, which allowed us to correlate biomass with CO_2 exchange on a per plot basis.

Microbial Respiration

We previously reported rates of potential CO_2 production (per gram dry weight soil) in a synthesis paper (Craft et al. 2003). These data are also reported in the present paper in order to contrast them with potential CO_2 production expressed on a per gram ash-free dry weight basis, and to make comparisons with potential CH_4 production. In July 1999, samples of the top 10 cm of soil were randomly collected from each created and natural marsh along two 30-m transects; one was the same transect used for the CO_2 exchange measurements, and the other was about 10 meters further inland.

At each site, sixteen soil cores were collected, eight from the front transect and eight from the back transect. Adjacent cores in each transect were paired and bulked to create four replicates per transect or eight per site. Soils were kept on ice until they were sieved (5.6-mm mesh) in a cold room within an N_2 atmosphere to remove large roots and shells. Forty-g wet-weight sediment samples were placed in 473-ml glass jars and sealed. Because anaerobic conditions dominate tidal marsh soils, the jars were filled with saline water (30 ppt salinity), leaving a 273-ml headspace, and maintained in an anaerobic atmosphere. Periodically, 10 ml of headspace gas was removed from each jar and replaced with industrial-grade N_2 to reduce concentrations of potentially toxic gases.

Jars were incubated in a temperature-controlled room at 25°C. CO_2 flux was measured on five dates over the course of a 76-day incubation and averaged. Soil respiration sampling, which began 14 days after collection, was measured as CO_2 accumulation in the jar headspace using a LI-COR 6200 Portable Infrared Gas Analyzer that was connected with Bev-a-Line tubing and operated in closed loop mode. The jar headspace was flushed with N_2 prior to sampling, and three separate flux measurements were made per jar. We used the minimum flux measurement of the three for our analysis to avoid artifacts that may have been introduced by connecting the IRGA. Periodically during the incubations, 50 ml of soil water was removed and replaced with new saline water to restore salinity and reduce the concentration of potential toxic substances.

Methane in the jar headspace was sampled through rubber septa 4–7 times over a period of 13 days, beginning 25 days after soil collection. The vast majority of the jars were sampled four times while a small number were incubated up to five days longer to improve the regression fit for the flux measurements. Ten ml of gas was removed with syringes and replaced with industrial-grade N_2. Methane concentration was determined using a Hewlett Packard 5890 gas chromatograph fitted with a flame ionization detector and Porapak Q column. Gas samples that were not immediately analyzed were refrigerated for a maximum of two days. Previous studies showed CH_4 concentrations decreased by less than 10% during storage using the same methods (Megonigal and Schlesinger 2002). Two days after collection, soils were dried at 105°C to constant weight, then combusted at 400°C for 16 hours to quantify soil organic matter (SOM) measured as % loss-on-ignition (LOI). The loss of carbon during the incubation was a negligible fraction of the SOM pool.

Statistical Analysis

A balanced design for within-marsh replication was used for field and lab experiments. The significance level was set at $\alpha = 0.05$ for all statistical tests. Statistical differences between created marsh and natural marsh pairs were assessed using the Wilcoxon exact test (Sokal and Rohlf 1981). Ordinary least squares regressions and correlations were performed to assess relationships among carbon cycle attributes and created marsh age. SAS statistical software was used to generate descriptive statistics and perform statistical tests (SAS 1990).

RESULTS

Plant Biomass

Two created marshes, Y1 and Y11, had significantly less aboveground biomass than their natural reference sites ($p < 0.04$), while two older created marshes, Y13 and Y28, had significantly more aboveground biomass ($p < 0.04$, Figure 1a). There were no significant differences in biomass on the remaining sites, which ranged in age from four to 27 years. These results generally agreed with data reported by Craft et al. (2003) for the same sites one year earlier, with two exceptions. They found that the 3-year-old created marsh had significantly less biomass than its paired natural marsh, whereas we found no significant difference between the two marshes a year later when the created marsh was 4 years old. In addition, they found no difference in aboveground biomass between the created 28-year-old site (Y28) and its natural marsh reference.

Field CO_2 Exchange

GPP was generally comparable in even the youngest of the created and natural marsh pairs (Figure 2a). The only exception was the Y11 site, which significantly under-performed its natural reference marsh in the first year (July 1998, $p = 0.02$). At 3 years of age, site Y3 had significantly lower GPP than its reference marsh (July 1998, $p = 0.02$), but at 4 years of age there was no significant difference between the sites (July 1999, $p = 0.84$). Natural marsh GPP was correlated with created marsh age (July 1999, $r^2 = 0.59$, $p = 0.03$, Figure 2b).

Marsh R was significantly greater in created marshes than natural marshes in several instances (Y8 and Y28 in October 1998; Y28 in January 1999; Y8 in July 1999; $p < 0.02$). R was significantly lower in the created marsh Y11 than its natural marsh in

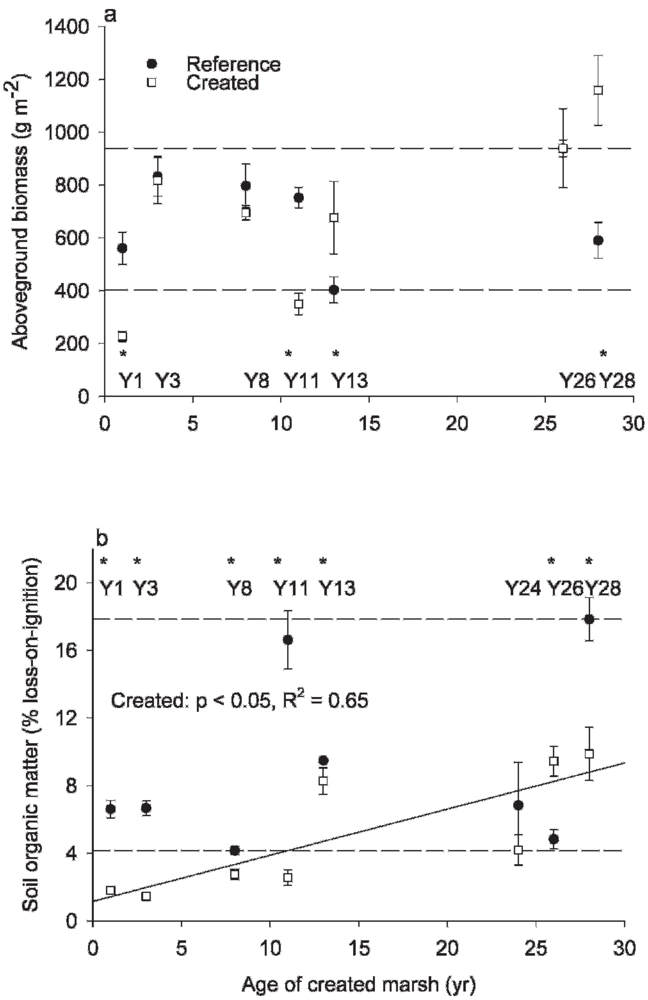


Figure 1. a) Marsh aboveground biomass and b) soil organic matter. Asterisks indicate that the created and paired reference marshes were significantly different ($p < 0.05$) based on Wilcoxon exact tests. Horizontal dashed lines represent the range of values measured in the reference marshes. Regression line illustrates the relationship between created marsh SOM and marsh age when one was found. All data are expressed as mean ± 1 SE.

July 1998 and January and March 1999 ($p < 0.04$). R in created marshes was related to marsh age in July 1998 and 1999 ($p < 0.04$, $r^2 = 0.54$ and 0.57, respectively, Figures 2c,d). However, R in natural marshes was also related to created marsh age in July 1999 ($r^2 = 0.61$, $p = 0.02$, Figure 2d). R in created marshes was correlated with SOM content in January 1999 only ($r = 0.78$, $p = 0.04$).

Although the overall significance of differences between created and natural marshes was not tested because of the chronosequence design, mean created marsh NEE was greater than natural reference marsh NEE in 79% of the cases (Figures 2e,f). This suggested a tendency for the carbon sequestration potential of created marshes to equal or exceed that of the natural marshes. However, there was no relationship between instantaneous NEE and created marsh age (Figures 2e,f). Created sites Y11 and Y28 had significantly greater NEE than their natural marsh reference in July 1998, and site Y13 had significantly greater NEE in July 1999 ($p < 0.04$).

Daily Integrated NEE

Q_{10} values were 1.6 at natural marsh Y26 and 1.4 at natural marsh Y28 during July 1999 sampling. We used the average of these values to model hourly and daily R. Although instantaneous CO_2 exchange was based on three to five plots (usually five) per site, daily CO_2 exchange could only be modeled for two plots per site. Therefore, comparing instantaneous and modeled gas exchange data directly between marsh pairs would be inappropriate. However, modeled GPP and R followed the same patterns as the instantaneous results when all marsh pairs were considered. Due to the variability in NEE and small sample sizes, there was no clear relationship between modeled and instantaneous NEE across all marsh pairs.

Daily integrated natural marsh GPP and R, which were measured and modeled independently, were significantly related ($p < 0.01$, adj. $r^2 = 0.81$). Old created marshes generally had higher daily R and GPP rates than the young created marshes, but the statistical differences could not be tested due to sample size limitations. There were no consistent patterns in NEE in comparisons of young (< 5 years) versus intermediate-aged (9–14 years) created marshes or intermediate versus old created marshes.

Microbial Respiration

On a soil dry weight basis, potential microbial respiration (CO_2 and CH_4) determined in lab incubations was significantly lower in created sites Y1, Y3, and Y11 than in the natural reference sites ($p < 0.01$; Figures 3a,b). Created marsh Y28 had lower potential CO_2 production than its reference marsh, a difference that was marginally significant ($p = 0.049$, Wilcoxon test, Figure 3a). Although it is a minor issue with respect to our data interpretation, Craft et al. (2003) analyzed the same data set using a paired t-test and found no significant difference between the Y28 marshes. Created site Y26 was the only marsh with significantly higher potential microbial CO_2 respiration than its natural reference site ($p < 0.01$). Potential microbial CO_2 respiration from created marsh soils was significantly related to marsh age ($p < 0.05$). Methane production repre-

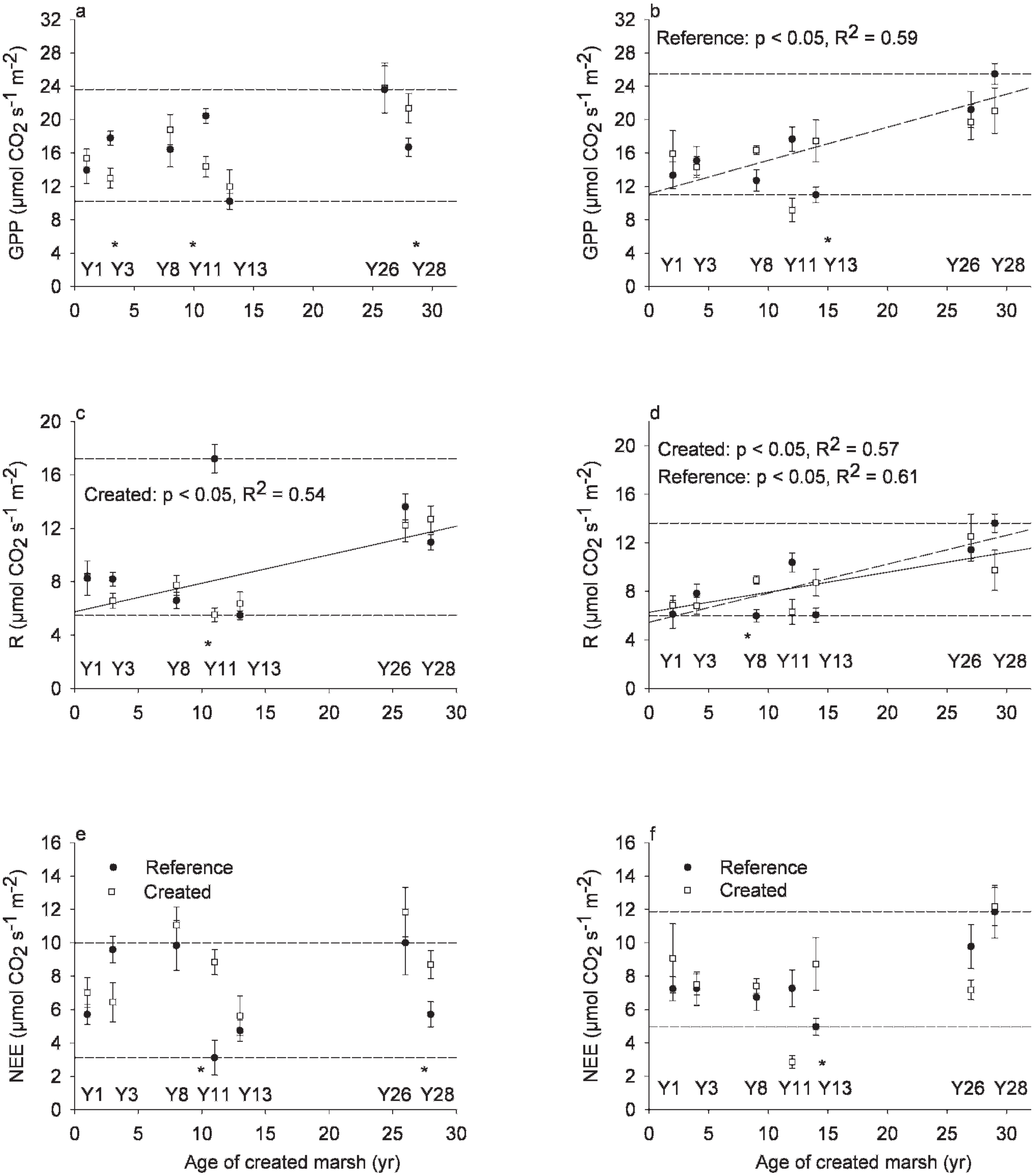


Figure 2. Gross primary productivity (a-1998, b-1999), respiration (c-1998, d-1999) and net ecosystem exchange (e-1998, f-1999) in July 1998 and 1999 versus created marsh age. Asterisks indicate that the created and paired reference marshes were significantly different ($p < 0.05$) based on Wilcoxon exact tests. Horizontal dashed lines represent the range of values measured in the natural marshes. Regression lines (created-solid, reference-dashed) indicate a significant relationship between the marsh component and created marsh age when one was found. Note that summer 1999 was hotter and drier than summer 1998. All data are expressed as mean ± 1 SE.

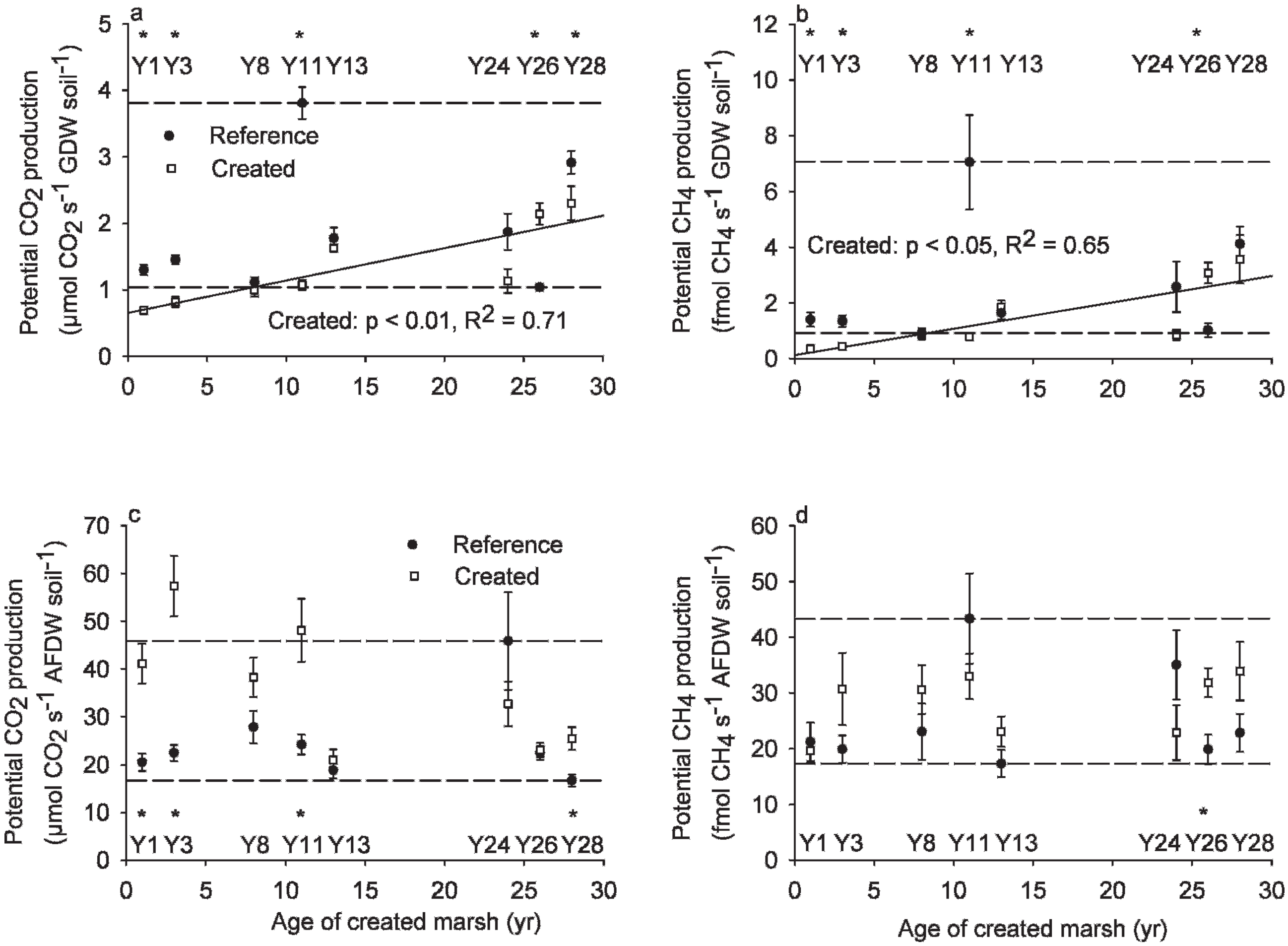


Figure 3. Potential CO_2 production (a-per GDW, c-per AFDW) and potential CH_4 production (b-per GDW, d-per AFDW). Asterisks indicate that the created and natural reference marshes were significantly different ($p < 0.05$) based on Wilcoxon exact tests. Horizontal dashed lines indicate the range of values measured in the reference marshes. Regression lines indicate a significant relationship between the created marsh component and created marsh age. Panel a is reprinted with permission from Craft et al. (2003). All data are expressed as mean ± 1 SE.

sented less than 1% of total microbial respiration in created and natural marshes as expected given high concentrations of sulfate in the incubation water. Microbial CO_2 respiration was correlated with aboveground biomass and *in situ* R in July 1999 ($p < 0.05$, $r > 0.76$).

On a carbon mass (i.e., ash-free dry weight) basis, all created marshes but Y24 and Y26 had higher potential CO_2 production than their natural reference marsh. CO_2 production in created marshes was significantly greater than in reference marshes for sites Y1, Y3, Y11, and Y28 (Figure 3c). Created site Y26 had a significantly higher potential CH_4 production rate than its natural reference marsh. SOM content (AFDW) explained a significant amount of the variation in potential CH_4 production in the created and natural marshes (adj. $r^2 = 0.85$, $p < 0.001$, adj. $r^2 = 0.59$, $p = 0.02$, respectively). There was no relationship between potential microbial CO_2 production and SOM content in either the created or natural marshes.

DISCUSSION

A conceptual model proposed by Craft et al. (2003) identified three distinct trajectories of ecosystem development in created salt marshes. In this model, plant biomass quickly reached parity with natural marshes, but other carbon pools, such as SOM, developed much more slowly (e.g., Seneca et al. 1985, Broome et al. 1986, Craft et al. 1999). Our process-level investigations suggest that key carbon fluxes develop rapidly in created salt marshes, essentially at a pace that matched, or perhaps exceeded, that of plant biomass.

Generally, created marsh aboveground biomass either matched or exceeded reference marsh aboveground biomass. The only exceptions were the youngest site (1 year) and the consistently underperforming Y11 marsh. In comparison to a study that occurred one year earlier in these same marshes, two of the created marshes showed significant improvement relative to the biomass of their reference marshes. Craft et al. (2003) determined that there was no difference between created and natural marsh aboveground biomass in Y28 (28 years old) and significantly less biomass in Y3 (3 years old). One year later, we found that Y28 had significantly more biomass than its reference marsh, and there was no significant difference in Y3. Assuming that the Y11 marsh was an outlier (see discussion following), we suggest that a minimum of three years is required before aboveground plant biomass will meet or exceed levels in natural marshes. Several previous studies have reported comparable rates of plant biomass development in created tidal marshes (Broome et al. 1982, Broome et al. 1988, Craft et al. 1999), while others suggested a somewhat longer time frame of between 5 and 15 years (Seneca et al. 1985, Broome et al. 1986, Langis et al. 1991, Craft et al. 2003). A relevant question is whether a minimum period of 3–5 years to reach reference levels of plant biomass is a constraint on the pace of CO_2 gas exchange development in created wetlands. In other words, how closely does ecosystem gas exchange track shoot biomass in developing tidal marshes?

Although the pace of development for both gas exchange and shoot biomass was rapid, there were differences in the trajectories of these attributes that suggest gas exchange may not strictly track shoot biomass. In fact, created marsh plant biomass did not correlate with GPP, R, or NEE (p-values ranged from 0.07 to 0.21). The youngest site (Y1) had significantly less shoot biomass than the natural reference site in 1998 (Craft et al. 2003) and 1999 (present study), yet there were no significant differences in GPP, R, or NEE. Some disparity can be expected between the gas exchange and shoot biomass data in these studies because they were measured at different times of year (July and October, respectively), and each represents different levels of temporal integration (instantaneous versus cumulative, respectively). Nonetheless, the present data suggest that it is possible for a created marsh to meet or exceed reference marsh gas exchange rates 1–2 years before shoot biomass becomes comparable. This could occur if plants at young sites ($<$ 3 years old) have higher leaf-level photosynthetic rates than in the natural marshes. Rather than allocating photosynthate to shoot production, plants in the Y1 marsh may have allocated energy to root production to mine N, P, or other nutrients from the nutrient-poor sandy substrate. We do not have sufficient data to evaluate this hypothesis, but it is notable that Y1 had significantly higher leaf N levels (1.23%, mean of 10 observations, unpublished analysis) than the other created sites in this study (range 0.68% to 0.99%). Leaf N content is generally positively correlated with leaf-level photosynthetic rates because Rubisco and chlorophyll, key biochemicals in the photosynthetic apparatus, are N enriched. Soil nutrient status might be an important regulator of this response because GPP tracked biomass more closely at site Y3 where leaf N levels were comparable to its paired natural site (0.80% and 0.71%, respectively). Testing this hypothesis would require closely following changes in plant physiology in the first few years following marsh creation, and perhaps simultaneously manipulating soil N availability.

The positive correlation between created marsh R and created marsh age would seem to suggest that the oldest created marshes were at an intermediate phase of succession nearly 30 years after marsh creation. However, because reference marsh GPP and R were also related to created marsh age, marsh location and marsh age might be confounded variables for these carbon cycle attributes. We cannot explain why location may have affected gas exchange rates, but possibilities include local nutrient sources and differences in solar exposure. There were no similar correlations between created marsh age and reference marsh plant biomass or soil organic matter. Thus, the appropriate comparisons for assessing change in gas exchange rates over time were between created and reference marsh pairs, and these data suggest that either gas exchange attributes develop very rapidly (i.e., $<$ 3 years) or the model proposed by Odum (1969) does not apply to created marshes. To assess early and intermediate successional patterns in field CO_2 exchange as described by Odum (1969), future research should focus on following young (1–5 years old) created marshes with frequent sampling.

Site Y11 was a consistent exception to the general pattern of rapid convergence of natural and created marsh carbon cycling. The created Y11 site had significantly lower aboveground biomass, SOM, GPP (July 1998), R (July 1998), and potential CH_4 and CO_2 production than its natural reference marsh. We attribute the site's poor performance primarily to flaws in design and construction. Unlike natural marshes that grade smoothly into upland, created marsh Y11 was bordered by a large sand berm that was upwards of 3 m higher than the adjacent marsh. It appeared that sand transport off the berm onto the marsh surface was stressing plants through burial and thereby limiting overall carbon cycling. Our supposition is in agreement with Craft et. al (2003) who observed significantly higher sedimentation rates in created Y11 than its natural reference marsh. Site Y11 emphasizes the importance of considering the adjacent upland system when designing created tidal marshes.

Potential microbial respiration rates were generally higher in created marshes than reference marshes when expressed as CO_2 production per gram of soil organic matter (Figure 3c). This suggests that heterotrophic microbes in the created marshes mineralized organic carbon more efficiently than those in the natural marshes. Because the incubation conditions were the same for all sites, differences in mineralization efficiency were due to soil properties such as nutrient availability or soil organic carbon quality. A difference in soil carbon quality is consistent with the fact that the lignin content of soil macro-organic matter, consisting largely of live and dead root material, was significantly higher in reference sites than created sites Y1, Y3, Y8, and Y13 (Craft et al. 2003). The influence of this age-related decline in soil carbon respiration efficiency (i.e., respiration per gram carbon) was offset by the increase in the size of the soil carbon pool because the overall soil organic matter mineralization rate (i.e., per gram soil) increased with age and soil organic carbon content (Figures 1b, 3a–d). Although it can take two decades to establish natural levels of soil carbon in created salt marshes, we show that most of the major carbon fluxes are established in less than five years.

Microbial respiration appeared to contribute less than root respiration to total CO_2 emissions from these created marsh soils. Despite the increase in soil organic matter mineralization rate (see previous paragraph), *in situ* summer R (i.e., microbial + root respiration) did not increase with age when compared to the reference marshes (Figures 2c,d). Only in January 1999, when plant respiration was minimal, was *in situ* R in the created marshes significantly related to SOM content (adj. $r^2 = 0.52$, $p = 0.04$). In addition, daily integrated rates of marsh R and GPP in the natural marshes were highly related ($p < 0.01$, adj. $r^2 = 0.81$) suggesting that labile plant compounds were the largest source of CO_2 measured in soil respiration. Most respiration of labile carbon would be expected to occur directly in the roots rather than by microbial respiration of rhizodeposits. The observation that SOM content (AFDW) explained a significant amount of the variation in potential CH_4 suggests that methanogens were carbon limited as expected in wetland soils (Megonigal et al. 2004).

In contrast to the 2–3 year time frame for aboveground biomass restoration, Craft et al. (2003) reported that more than 28 years was required before created marsh SOM content reached levels that met or exceeded the natural marshes on the same chronosequence. Several processes have contributed to the linear increase in SOM with marsh age that they reported. Initially, plant biomass increases, which injects root carbon into the soil profile and deposits shoot carbon on the soil surface. Increasing shoot biomass favors deposition of fine suspended sediments (Darke and Megonigal 2003) and associated particulate carbon. As soil elevation increases due to sediment deposition on the soil surface, flooding frequency and associated sediment inputs decline (Morris et al. 2002, Darke and Megonigal 2003), resulting in less dilution of carbon inputs by sediment. We did not evaluate whether the created marshes were accumulating soil organic carbon at the same rate as natural marshes because surface accretion rates were not measured and radionuclide dating was not possible on created sites. However, we speculate that accretion was more rapid at the young created sites because of their relative low elevations compared with natural or old created sites (e.g., Ward et al. 2003). Craft et al. (2003) reported that sedimentation rates were significantly higher in the created Y1 and Y11 marshes than their paired natural marshes, but there were no corresponding significant differences for created marshes aged 24–28 years (rates were not measured at Y3, Y8, or Y13). Thus, the capacity of created and natural marshes to sequester carbon may be similar at ages less than 30 years despite lower soil carbon concentrations at the created sites.

As with all space-for-time substitution studies, we assumed that differences caused by age would be greater than differences caused by other sources of variability. This assumption held in some instances (e.g., potential CO_2 mineralization, Figures 3a,b) where the response variable was significantly related to created-site age across the created sites, but not the natural sites. In other cases, this assumption did not hold (e.g., July 1999 R, Figure 2d), and interpretation depended on comparisons between a pair of created and natural plots at the same location. The paired-site-comparison approach presumably accounted for variation among sites caused by location-specific environmental factors such as salinity (Table 1). However, there could also be stochastic influences on ecosystem development related to initial conditions that are marsh-specific (Haltner et al. 1997, Boyer et al. 2000, Zedler and Callaway 2000), which a paired-site approach would not capture. All of these issues can be avoided by tracking ecosystem development on replicate sites in real time, an approach that is feasible for these low-stature ecosystems (e.g., Craft et al. 1999).

In conclusion, aboveground biomass in created and natural salt marshes in North Carolina reached parity within 2 to 3 years unless the site was improperly constructed. Plant productivity at one site (Y11) seemed to be under stress due to constant burial from an upland source of sand. GPP, R, and NEE developed at a pace similar to plant biomass development. The possibility that GPP, R, and NEE may develop more rapidly than plant biomass deserves attention from ecophysiologists because it suggests there are feedbacks between changes in plant physiology and ecosystem processes that have been overlooked in created marsh research. Further studies of soil elevation change with age are required to understand feedbacks between elevation, sedimentation, and carbon cycle development.

ACKNOWLEDGMENTS

We acknowledge the kind support of Donald Kelso and capable assistance from Joanna Cornell, Milena Arciszewski, Kristin Fitzgerald, Andy May, Carrie deJaco, Cheryl Vann, Bill Kornicker, Shamus Goss, and Katherine Connors. This research was supported by a grant from the U.S. Environmental Protection Agency's Science To Achieve Results (STAR) program through grant #826111-01-0. Although the research described in the article has been funded wholly or in part by the U.S. Environmental Protection Agency's STAR program through grant #826111-01-0, it has not been subjected to EPA review and therefore does not necessarily reflect the views of the Agency, and no official endorsement should be inferred.

LITERATURE CITED

Boyer, K. E., J. C. Callaway, and J. B. Zedler. 2000. Evaluating the progress of restored cordgrass (*Spartina foliosa*) marshes: belowground biomass and tissue nitrogen. Estuaries 23(5): 711–21.

Broome, S. W., E. D. Seneca, and W. W. Woodhouse Jr. 1982. Establishing brackish marshes on graded upland sites in North Carolina. Wetlands 2:152–78.

Broome, S. W., E. D. Seneca, and W. W. Woodhouse Jr. 1986. Long-term growth and development of transplants of the salt-marsh grass *Spartina alterniflora*. Estuaries 9:63–74.

Broome, S. W., E. D. Seneca, and W. W. Woodhouse Jr. 1988. Tidal salt marsh restoration. Aquatic Botany 32:1–22.

Conn, C. E. and F. P. Day Jr. 1997. Root decomposition across a barrier island chronosequence: litter quality and environmental controls. Plant and Soil 195:351–64.

Craft, C. B. 1997. Dynamics of nitrogen and phosphorous retention during wetland ecosystem succession. Wetlands Ecology and Management 4:177–87.

Craft, C. B., S. W. Broome, and E. D. Seneca. 1988a. Nitrogen, phosphorous, and organic carbon pools in natural and transplanted marsh soils. Estuaries 11:272–80.

Craft, C. B., S. W. Broome, E. D. Seneca, and W. J. Showers. 1988b. Estimating sources of soil organic matter in natural and transplanted estuarine marshes using stable isotopes of carbon and nitrogen. Estuarine, Coastal and Shelf Science 26:633–41.

Craft, C. B., S. W. Broome, and E. D. Seneca. 1989. Exchange of nitrogen, phosphorous, and organic carbon between transplanted marshes and estuarine waters. Journal of Environmental Quality 18:206–11.

Craft, C., J. Reader, J. N. Sacco, and S. W. Broome. 1999. Twenty-five years of ecosystem development of constructed *Spartina alterniflora* (Loisel) marshes. Ecological Applications 9:1405–19.

Craft, C., P. Megonigal, S. Broome, J. Stevenson, R. Freese, J. Cornell, L. Zheng, and J. Sacco. 2003. The pace of ecosystem development of constructed *Spartina alterniflora* marshes. Ecological Applications 13:1417–32.

Darke, A. K. and J. P. Megonigal. 2003. Control of sediment deposition rates in two mid-Atlantic coast tidal freshwater wetlands. Estuarine, Coastal and Shelf Science 57:255–68.

Dayton, P. K., L. A. Levin, T. S. Talley, A. McCray, and A. R. Bustamante. 1996. Early successional measurements in a restored tidal marsh. p. 197–202. Scripps Institution Oceanography, University of California, San Diego, CA, USA.

Haltiner, J., J. B. Zedler, K. E. Boyer, G. D. Williams, and J. C. Callaway. 1997. Influence of physical processes on the design, functioning and evolution of restored tidal wetlands in California (USA). Wetlands Ecology and Management 4:273–91.

Langis, R., M. Zalejko, and J. B. Zedler. 1991. Nitrogen assessments in a constructed and a natural salt marsh of San Diego Bay. Ecological Applications 1:40–51.

LaSalle, M. W., M. C. Landin, and J. G. Sims. 1991. Evaluation of the flora and fauna of a *Spartina alterniflora* marsh established on dredged material in Winyah Bay, South Carolina. Wetlands 11:191–208.

Megonigal, J. P. and W. H. Schlesinger. 2002. Methane-limited methanotrophy in tidal freshwater swamps. Global Biogeochemical Cycles 16(4), 1088, doi: 10.1029/2001GB001594.

Morris, J. T., P. V. Sundareshwar, C. T. Nietch, B. Kjerfve, and D. R. Cahoon. 2002. Responses of coastal wetlands to rising sea level. Ecology 83:2869–77.

Neubauer, S. C., W. D. Miller, and I. C. Anderson. 2000. Carbon cycling in a tidal freshwater marsh ecosystem: a carbon gas flux study. Marine Ecology Progress Series 199:13–30.

Noon, K. F. 1996. A model of created wetland primary succession. Landscape and Urban Planning 34:97–123.

Odum, E. P. 1969. The strategy of ecosystem development. Science 164:262–70.

Padgett, D. E. and J. L. Brown. 1999. Effects of drainage and soil organic content on growth of *Spartina alterniflora* (Poaceae) in an artificial salt marsh mesocosm. American Journal of Botany 86:697–702.

Poach, M. E. and S. P. Faulkner. 1998. Soil phosphorous characteristics of created and natural wetlands in the Atchafalaya Delta, LA. Estuarine, Coastal and Shelf Sciences 46:195–203.

Radford, A. E., H. E. Ahles, and C. R. Bell. 1968. Manual of the vascular flora of the Carolinas. The University of North Carolina Press, Chapel Hill, NC, USA.

SAS. 1990. SAS/STAT User's Guide, version 6. SAS Institute, Cary, NC, USA.

Segel, I. H. 1976. Biochemical Calculations: How to Solve Mathematical Problems in General Biochemistry. John Wiley & Sons, Inc., New York, NY, USA.

Seneca, E. D., S. W. Broome, and W. W. Woodhouse Jr. 1985. Comparison of *Spartina alterniflora* Loisel. transplants from different locations in a man-initiated marsh in North Carolina. Wetlands 5:181–90.

Shafer, D. J. and W. J. Streever. 2000. A comparison of 28 natural and dredged material salt marshes in Texas with an emphasis on geomorphological variables. Wetlands Ecology and Management 8:353–66.

Sokal, R. R. and F. J. Rohlf. 1981. Biometry, second edition. W. H. Freeman and Company, San Francisco, CA, USA.
Streever, W. J. 2000. *Spartina alterniflora* marshes on dredged material: a critical review of the ongoing debate over success. Wetlands Ecology and Management 8:295–316.
Ward, K. M., J. C. Callaway, and J. B. Zedler. 2003. Episodic colonization of an intertidal mudflat by native cordgrass (*Spartina foliosa*) at Tijuana Estuary. Estuaries 26(1):116–30.
White, D. S. and B. L. Howes. 1994. Nitrogen incorporation into decomposing litter of *Spartina alterniflora*. Limnology and Oceanography 39:133–40.
Zedler, J. B. and J. C. Callaway. 2000. Evaluating the progress of engineered tidal wetlands. Ecological Engineering 15(3–4):211–25.

Manuscript received 6 March 2006; revision received 27 November 2006; accepted 23 January 2007.

Style Manual Website:

https://www.agronomy.org/publications/style/